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Contents

MODERNIZATION

User Needs Assessment for the Modernization of the Forest Vegetation Simulator	2
<i>Wade T. Tinkham, Linda M. Nagel, and Molly Roske</i>	
Development of a New Interface for the Forest Vegetation Simulator.....	4
<i>Nicholas Crookston and Michael A. Shettles</i>	

DEVELOPMENT OF NEW VARIANTS

Development of the Organon Based fvs Variants, Organon Pacific Northwest (op) and Organon Southwest (oc)	8
<i>Erin Smith-Mateja</i>	
The Acadian Variant of the Forest Vegetation Simulator: Continued Development and Evaluation.....	10
<i>Aaron Weiskittel, John Kershaw, Nicholas Crookston, and Chris Hennigar</i>	
Development and Evaluation of an Individual Tree Growth and Yield Model for the Adirondacks Region of New York	14
<i>Aaron Weiskittel, Christian Kuehne, John Paul McTague, and Mike Oppenheimer</i>	
Modeling Future Dynamics of European Beech Forests for Ground Beetle Conservation	18
<i>Giorgio Vacchiano, Roberta Berretti, Elena Regazzoni, Flavio Ruffinatto, and Matteo Negro</i>	

DEVELOPMENT OF EXTENSIONS, POST PROCESSORS, AND LINKS TO OTHER MODELS

The FVS-WRENSS Water Yield Post-Processor: Validation of Snow-Dominated Procedures	24
<i>Robert N. Havis</i>	
Linking FVS and TELSA via the API.....	34
<i>Donald C.E. Robinson and Sarah J. Beukema</i>	
Linking FVS to 3D Fire Models: Introduction to STANDFIRE, a Platform for Stand-Scale Fuel Treatment Analysis.....	37
<i>Russ Parsons, Lucas Wells, Francois Pimont, W. Matt Jolly, Greg Cohn, Rod Linn, Ruddy Mell, and Chad Hoffman</i>	
A Framework for Evaluating Forest Restoration Alternatives and their Outcomes, Over Time, to Inform Monitoring: Bioregional Inventory Originated Simulation Under Management.....	40
<i>Jeremy S. Fried, Theresa B. Jain, Sara Lorenzo, Robert F. Keefe, and Conor K. Bell</i>	

COMPUTATIONAL TECHNIQUES

An Evaluation of CLIMATE Site Index in Large-Tree Diameter Growth Modeling of Selected Tree Species in the Great Lakes Region, U.S.A.....	52
---	----

Ram K. Deo, Robert E. Froese, Matthew B. Russell, and Michael J. Falkowski

Adjusting Canopy Cover Estimates for Non-Random Spatial Distributions in FVS	57
--	----

Michael Shettles and Erin Smith-Mateja

Live Tree Carbon Stock Equivalence of Fire and Fuels Extension to the Forest Vegetation Simulator and Forest Inventory and Analysis Approaches	60
--	----

James E. Smith and Coeli M. Hoover

Theoretical Foundation of Stage's Formulation of Stand Density Index	64
--	----

Hsien-chih Bryan Lu, Fred Martin, and Ralph Johnson

REGENERATION MODELING

Modeling the Impact of Overstory Density on the Regeneration Dynamics of Missouri Ozark Forests	72
---	----

Lance A. Vickers, David R. Larsen, Benjamin O. Knapp, Daniel C. Dey, and John M. Kabrick

Development and Assessment of Regeneration Imputation Models for National Forests in Oregon and Washington.....	74
---	----

Karin M. Kralicek, Andrew Sánchez Meador, and Leah C. Rathbun

EVALUATION OF BASE MODEL

Performance of FVS Variants in Relation to an Extensive Chronosequence and Remeasurement Dataset for Eastern White Pine (<i>Pinus strobus</i> , L.) in Central Maine.....	78
--	----

David Ray and Robert Seymour

Using Forest Vegetation Simulator (FVS) to Calculate Cover Type Transition Probabilities of Deferred/Altered Stands Within the Border Lakes Subsection	82
--	----

Curtis L. VanderSchaaf

Evaluating Diameter Increment in Disturbed Forests Across the U.S. Lake States	87
--	----

Macklin Glasby and Matthew Russell

Evaluation of the FVS-CR Diameter Growth Model and Potential Modifications in Structurally-Complex Ponderosa Pine Forests.....	89
--	----

Yvette L. Dickinson, Michael A. Battaglia, and Lance A. Asherin

EVALUATION OF THE FIRE AND FUELS EXTENSION

Evaluation of the Fire and Fuels Extension to the Forest Vegetation Simulator Within the Missouri Ozarks.....	94
<i>Casey R. Ghilardi, Benjamin O. Knapp, Hong S. He, David R. Larsen, and John M. Kabrick</i>	
Validation and Development of Postfire Mortality Models for Upland Forest Tree Species in the Southeastern United States.....	98
<i>Tara L. Keyser, Virginia L. McDaniel, Robert N. Klein, Dan G. Drees, Jesse A. Burton, and Melissa M. Forder</i>	
Estimating Canopy Bulk Density and Canopy Base Height for Conifer Stands in the Interior Western United States Using the Forest Vegetation Simulator Fire and Fuels Extension.....	110
<i>Seth Ex, Frederick (Skip) Smith, Tara Keyser, and Stephanie Rebain</i>	
Sensitivity of Crown Fire Modeling to Inventory Parameter Dubbing in FVS.....	114
<i>Wade T. Tinkham, Chad M. Hoffman, Seth A. Ex, Michael A. Battaglia, and Alistair M.S. Smith</i>	

FIRE, CARBON, AND CLIMATE PROJECTS

Using Climate-FVS to Inform Management Decisions: Three Case Studies from the American Southwest	126
<i>Andrew Sánchez Meador, Alicia Azpeleta, Michael Stoddard, Benjamin Bagdon, and Sushil Nepal</i>	
Integrating Large Wildfire Simulation and Forest Growth Modeling for Restoration Planning	129
<i>Alan A. Ager, Rachel Houtman, Robert Seli, Michelle A. Day, and John Bailey</i>	

FOREST HEALTH PROJECTS

Use of the Forest Vegetation Simulator and the Southern Pine Beetle Event Monitor to Identify Silvicultural Treatments for the Reduction of Southern Pine Beetle Hazard and Enhancement of Restoration on the North Carolina Piedmont	140
<i>Jason A. Rodrigue, Chad. E. Keyser, and John T. Nowak</i>	
Estimating Changes to Forest Structure as a Result of Forest Pests: Using FVS to Simulate Potential Effects of Emerald Ash Borer Across a Broad Landscape	149
<i>Andrew J. McMahan and William B. Monahan</i>	

FSVEG SPATIAL DATA ANALYZER PROJECTS

Using Landfire, FSveg Spatial Data Analyzer Nearest Neighbor, Forest Vegetation Simulator, and FlamMap to Compare Treatment Effects Across a Landscape.....	154
<i>James Arciniega</i>	
Spatial Modeling of Timber Ecosystem Services: Linking the FVS Econ Extension and FSveg Spatial Data Analyzer to Map Stumpage Value.....	164
<i>Christopher Haberland and Jonathan Marston</i>	

ECONOMICS

Economic Returns of White Spruce Plantation Thinning Scenarios Using Forest Vegetation Simulator (FVS).....	172
<i>Curtis L. VanderSchaaf, Gordon Holley, and Joshua Adams</i>	
Comparing Unthinned Slash Pine Plantation Yield Predictions From Time-of-Planting	182
<i>Curtis L. VanderSchaaf, Gordon Holley, and Joshua Adams</i>	
Even- and Uneven-Aged Management Scenarios for Maximizing Economic Return in the Sweetgum-Nuttall Oak-Willow Oak Bottomland Hardwood Forest Types in the Lower Mississippi Alluvial Valley	190
<i>Sunil Nepal, Brent R. Frey, and James E. Henderson</i>	
Index of Authors.....	200

Modernization

User Needs Assessment for the Modernization of the Forest Vegetation Simulator

Wade T. Tinkham, Linda M. Nagel, and Molly Roske¹

The Forest Vegetation Simulator (FVS) is a widely used growth and yield modeling platform within the United States for assessing how vegetation will respond to natural succession, disturbances, and proposed management actions. FVS and all its extensions are the product of nearly 45 years of biometrics research. The science that underpins the model continues to undergo periodic advances as U.S. Department of Agriculture Forest Service (USDA) and university researchers improve our understanding of forest growth. However, the Suppose graphical user interface, which allows users a point-and-click way of generating simulations, has seen only limited updates over the last 20 years. The USDA Forest Service FVS Group has undertaken a multi-year process of modernizing FVS so that it can meet the computational and performance demands of modern forest management. This process is seeking to address issues with how users input data to the model, interact with the model through the Suppose platform, and to improve the usability of model outputs. The intention of the user needs assessment for the modernization process is to obtain user input to inform which components of the model require and are feasible for modernization to meet the 21st century demands that natural resource management will place on FVS.

In order to assess the range of model user needs, a mixture of social science techniques were utilized to seek input across a wide range of FVS users. These instruments were broadly structured around three elements of the typical user workflow with the model: (1) data input, (2) use of the Suppose interface, and (3) model outputs. The first

instrument consisted of semi-structured interviews (n=15 interviews) seeking input to identify the areas of greatest need for modernization within FVS. The interviews consisted of 21 predefined questions with a set of possible follow-up questions depending on the interviewee's initial response. These interviews were conducted with a targeted group of managers, researchers, and model developers that range in experience from 3 to 15 years of using the model. Throughout the interview and analysis process the identities of the interviewees remained anonymous from the FVS Group staff. These qualitative interviews provided a depth and richness to our understanding of what users view as the components in greatest need of modernization.

Once the interviews were conducted and transcribed, the interview transcripts were analyzed to determine any common themes that frequently arose. Following this summarization process, the FVS Group staff conducted an initial filtering of ideas for feasibility. This filtering resulted in 20 common themes or ideas arising from the interview process. A questionnaire of potential action items for modernizing FVS was created from this set of 20 ideas. The questionnaire was organized so that the user-identified need was stated, a potential solution was given, and then questionnaire recipients were asked to provide feedback or an alternative to the given solution. Respondents were also asked to rank the priority of each action item. An electronic questionnaire was sent to 32 model users representing a range of managers, researchers, and model developers, with 24 users responding. The written questionnaire was the chosen tool for engaging this group of participants to provide them

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with ample time for detailed feedback regarding the potential improvements and approaches this modernization process may undertake.

The range in user responses and ideas during this process reflects the wide-ranging applications for which FVS is utilized. Most users supported increasing the flexibility of how data is read into FVS, including adding other file formats (e.g., .xls, .xlsx, and .csv) and allowing FVS to connect to different database structures. Along with this, most users supported creating a utility that could translate different column headings to ensure data was formatted correctly. Across both data input and running simulations within the Suppose user interface, users supported the idea of improving how warning and error messages are issued. These warnings and errors might include an inventory data quality review for things like species codes or realistic tree structural parameters when data is first read into FVS, or this could include a report and graphics of stand structural metrics at the inventory time step to ensure the simulation is starting correctly. Finally, users would like to see improved

simulation data reporting capabilities beyond the current text and database link outputs. Part of these reporting capabilities might include advancing FVS to allow data outputs as .csv, .xlsx, or .xls formats and include the ability to output basic graphics as .pdf, .bmp, .jpeg, and .png formats.

Across most of the user suggested and supported ideas, building in a greater ability for users to customize how they interact with the model was generally desired. This could mean different aspects of the model could be turned off or hidden to create a simpler version of the model for novice users, with the option to turn on different utilities as their capabilities and needs advance. Additionally, these customizable features could allow users to adjust and save default settings for model functions like how data is read in, how keywords are parameterized, and how simulations are saved. Across the range of users, there is a general excitement to see this modernization of a tool they all greatly value and want to continue using in the modern world of forestry.

Development of a New Interface for the Forest Vegetation Simulator

Nicholas Crookston and Michael A. Shettles¹

Suppose (Crookston 1997) was the first graphical user interface to the Forest Vegetation Simulator (FVS). This paper presents FVSONline, a successor to the Suppose program, that does most of what Suppose does while providing much more functionality and a code base that is more sustainable and changeable than the Suppose program.

FVSONline builds FVS runs, makes the runs using FVS, and provides tools to compare the outputs in tabular and graphical formats. FVS runs are a collection of simulations each made of a collection of stands run under simulation alternatives. A collection of runs make up a project. Projects are stored on the computer system in a specific directory. Each project contains a copy of the software used to run the project and the user has control over if and when that software is updated. This allows users to freeze the models used to run a given analyses. All the input data for a project are stored in the same input database and all the outputs are stored in a single output database. The FVSONline tools for managing the input database include importing data from Microsoft Access, adding data from comma delimited files (.csv), and directly adding or editing data from the keyboard. The database system used by FVSONline is SQLite3, which is the most widely deployed database engine in the world today (thanks to D. Richard Hipp, see <http://www.hwaci.com/sw/sqlite>, accessed on April 3, 2017). The output database is automatically updated when a run is repeated ensuring that only up-to-date information is used in subsequent analyses steps. FVSONline supports importing and using existing FVS keyword component files (also known as .kcp or addfiles). It has been tested on large (ca 20,000 stands) runs and performs well. Tools for selecting and processing

individual stands in the input and output are included.

A goal of the FVSONline system is to provide for most of the analyses needs within the single package and thereby replace most of the post-processors (<https://www.fs.fed.us/fmnc/fvs/software/postprocessors.php>) that are currently being maintained and available only on Windows operating systems. Currently, the system automatically builds composite yield as well as stand and stock tables. If needed, specific output data can be easily selected and exported as .csv files. Under current development is a stand visualization tool fully linked in FVSONline that is similar to the Stand Visualization Tool (SVS, McGaughey 1997).

FVSONline is open-source, public-domain, and cross-platform (see <https://sourceforge.net/p/open-fvs/wiki/FVSONline>, accessed on April 3, 2017). It has been run under Ubuntu Linux, MACOSX, and Windows operating systems. The original requirements for FVSONline identified the need to provide a web-based system alleviating the need for users to install the software which is sometimes institutionally blocked, or owning powerful enough computers to do their analyses. However, some organizations have institutional constraints that restrict web-based services so it was clear that the software must also be usable to run locally on user's PC. FVSONline, therefore, can be run as FVSONlocal to meet this need (fig. 1).

Another goal was to provide a user interface that other programmers can contribute to without knowing C++ and both knowing and owning an expensive license to the specialized windowing management software on which Suppose is based.

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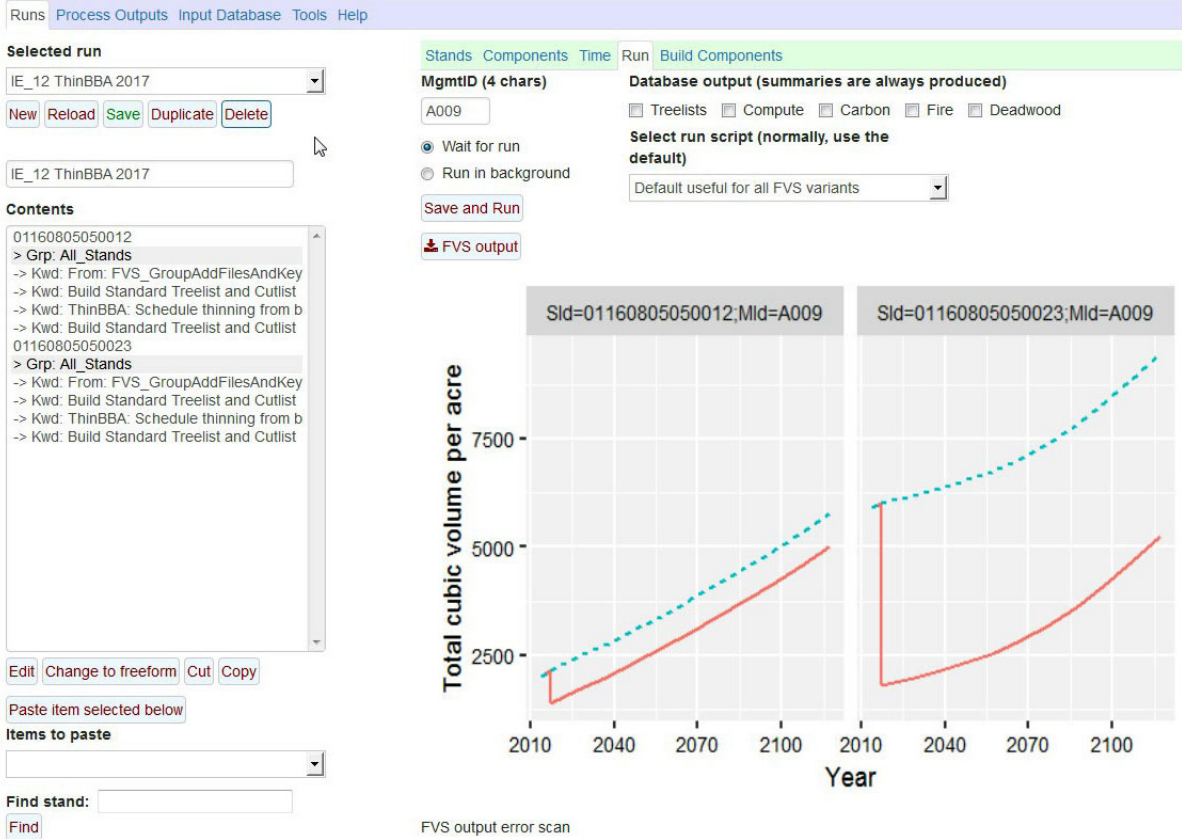


Figure 1—A view of the graphical confirmation of run success the user is given immediately after a run has completed. Note the different tab bars and individual tabs. Each reveal contents for building different pieces into a run and are aligned in a hierarchically intuitive fashion. This run was performed locally on a Windows 7 machine.

These issues have proven to be an impediment to making enhancements to Suppose over its entire lifetime of use. These key requirements were met by implementing the software in the R programming language (R Core Team 2016) making heavy use of the R package *shiny* (Chang and others 2017). When run as FVSONline, users only need a fully featured web browser. When run as FVSONlocal, R and several supporting packages as well as FVS are also required to be loaded to the user's PC. Besides being freely available, R has the advantage of being a scripted language with highly efficient mathematics, statistics, spatial analyses, graphics, and database tools—qualities that make FVSONline easy to extend and modify. This also allowed for the integration of the package *ggplot2*, a graphical system that allows users to easily create detailed multi-faceted quality graphics (fig. 2). The database

system used by FVSONline, SQLite3, is integrated into R and can be read and written by FVS through ODBC connections (<http://www.ch-werner.de/sqliteodbc>, accessed on April 7, 2017).

The open source, public domain and cross platform nature of R, including all availability of R packages made it a great tool for building FVSONline. Another reason R was chosen was that FVSONline supports the use of rFVS (<https://sourceforge.net/p/open-fvs/wiki/rFVS/>), a set of R functions that dynamically interact with FVS, providing the capability to include alternative growth, mortality, volume, regeneration establishment estimates, or any of these in combination as part of the FVS simulations. This system was used to implement the Acadian and Adirondacks models presented in these proceedings (Weiskittel and others 2017a, 2017b).

Diameter Distribution for Two Alternatives (2017, 2057)

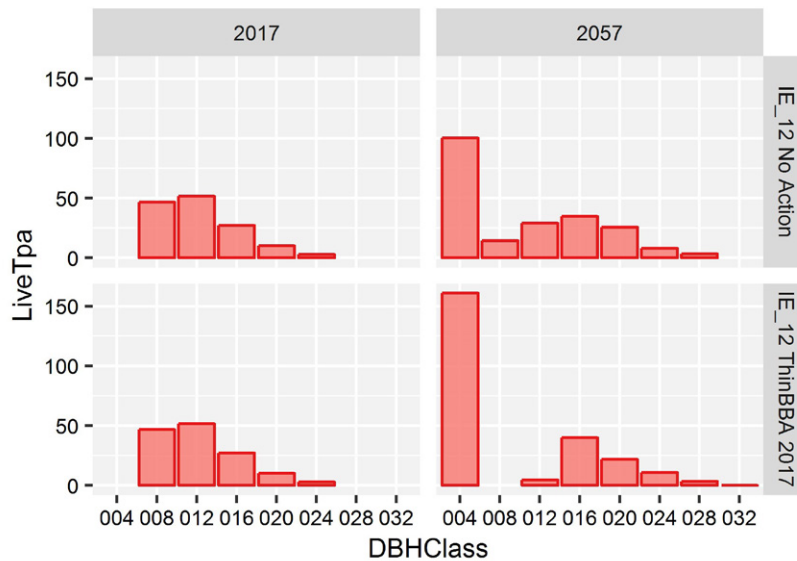


Figure 2—A graphical comparison of the differences in projected diameter distributions for no-thin and thin from below management actions. This was created using the *ggplot2* graphing capabilities built into FVSONline.

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Development of New Variants

Development of the Organon Based FVS Variants, Organon Pacific Northwest (OP) and Organon Southwest (OC)

Erin Smith-Mateja¹

ORGANON (ORegon Growth ANalysis and projectiON) is an individual-tree distance independent growth and yield model developed at Oregon State University (Hann 2011, Larsen Hann 1985, Ritchie Hann 1984). There are four versions of ORGANON that model growth for specific geographic areas or forest types. They include: (1) the Southwest Oregon version (SWO-ORGANON), (2) Northwest Oregon version (NWO-ORGANON), (3) the Stand Management Coop Version (SMC-ORGANON) specifically built for Northwest industrial landowners for shorter rotations, and (4) the Red Alder Plantation (RAP-ORGANON) built specifically for red alder plantations in the Northwest. ORGANON is used in the Pacific Northwest by the Bureau of Land Management (BLM), industrial landowners, consultants and universities. The model has a long development history, published equations and has been vetted among users in the Pacific Northwest. Oregon State University, College of Forestry Website provides two versions of ORGANON, a Disk Operating System (DOS) version and Dynamic Link Library (DLL) version. The DOS version is cumbersome and only allows for single stand processing. Because of this, consultants and companies have built their own interface that initiates the DOS executable or DLLs. In addition, ORGANON does not have growth equations for trees < 4.5 feet tall and has poor error report processing. This additional overhead can make the use of ORGANON frustrating and time consuming for the small landowner, small consultant, or public agencies. Through an interagency agreement with BLM, the U.S. Department of Agriculture, Forest Service has incorporated the ORGANON equations of the SWO, NWO and SMC versions of ORGANON into

two new versions of the Forest Vegetation Simulator (FVS), ORGANON Pacific Northwest (OP) variant (Smith-Mateja 2015b) and ORGANON Southwest (OC) variant (Smith-Mateja 2015a).

Early attempts to embed the ORGANON DLLs into FVS (Hamann 2012) did not provide the robustness desired, due to the inability to debug the DLL portion of the code and resolve internal errors. Soon after the initial attempt to make FVS run the DLL, David Hann released the DLL source code publicly (<http://www.cof.orst.edu/cof/fr/research/organon/downld.htm>). This made it possible to incorporate the FORTRAN code into the FVS base code.

To develop OP and OC, the ORGANON code was inserted into existing FVS variants. OP uses the base Pacific Northwest Coast (PN) variant (Keyser 2008a) code, and allows the user to choose the NWO or SMC versions. OP will check at each cycle for a valid ORGANON tree record, based on species (specified in the OP Variant Overview) and size (> 4.5 feet tall). If the tree is considered a valid tree, OP will simulate tree growth using the ORGANON equations. If the tree does not meet the requirements of a valid ORGANON tree, OP will simulate growth using the base PN equations. A similar process is used in OC, however OC is based on the Inland California and Southern Cascades (CA) variant code (Keyser 2008b) and embeds the ORGANON SWO equations.

Embedding the ORGANON code into an FVS variant of similar geographic coverage provides the opportunity to use the FVS extensions that are available to the base model. Specifically, embedding ORGANON into FVS will permit the use of the

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Fire and Fuels Extension, which is widely used for fuels management and carbon estimates in western Oregon. The inclusion of ORGANON within the overall FVS framework also provides additional FVS tools such as the use of more complex management options, the Event Monitor, FVS Extensions, and the use of the FVS Suppose or Online User Interface.

OC and OP variant executables and documentation are available on the FVS Website and are part of the suite of software included in the FVS setup program. Future development and enhancements will focus on improving the reporting of ORGANON calibration values and ORGANON related errors and warnings.

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The Acadian Variant of the Forest Vegetation Simulator: Continued Development and Evaluation

Aaron Weiskittel, John Kershaw, Nicholas Crookston, and Chris Hennigar¹

The Acadian Region of Maine and the Maritime Provinces of Canada are characterized by extensively managed, naturally-regenerated forest stands comprised of mixed species and multi-cohort structures. This area is quite distinct when compared to the rest of the Northeastern United States. This is because of a long history of varied management, it is the transition zone between the hardwood forests of the temperate zone and the softwood forest of the boreal zone, and there are over 25 commercial species present. A complex of topography, soil parent material, and climatic zones also creates a diverse forest. The Northeast Variant of the Forest Vegetation Simulator (NE) has a long history and covers a broad geographic region with previous testing indicating potential shortcomings in the Acadian Region, particularly for the spruce-fir (*Picea-Abies*) forest type (Bataineh and others 2013, Saunders and others 2008). Since 2008, efforts in Maine and New Brunswick have focused on developing an FVS variant specific to the Acadian Region (ACD).

Most of underlying equations for ACD have been developed and presented elsewhere (table 1). Like NE, ACD is an individual tree model with species-specific equations for crown width, total height, height to crown base, diameter and height increment, and mortality. Unlike NE, these equations developed using a novel annualization process (Weiskittel and others 2007), are influenced by climate site index based on latitude/longitude rather than relying on a user-supplied value, and do not use a potential times modifier approach, which can be problematic for equation development and application (Russell and others 2014). When

compared to NE across a range of forest types typical for the region, ACD stand net growth is often lower than NE even when ingrowth is incorporated (fig. 1). The differences are more apparent for hardwood and mixedwood stands with more limited differences in softwood stands. This is logical since most hardwoods are near their northern limit in Maine and would likely have slower growth than other portions in the Northeast. In addition, the shape of basal area yields over time are quite distinct between ACD and NE with ACD showing periods of stagnation and even decline, while NE predictions are very linear until reaching an asymptotic value that is maintained. This trend is also expected since the high natural regeneration densities in Maine can cause an extended self-thinning period for many stands. Continual model assessment for both ACD and NE will be conducted using long-term permanent plots and the U.S. Forest Service Forest Inventory and Analysis inventory in Maine.

Since the development of the baseline equations, additional model enhancements have continued and have primarily focused on modifying model behavior for certain conditions. In particular, this has focused on modifying spruce-fir growth and mortality for commercial thinning and spruce budworm [*Choristoneura fumiferana* (Clemens)] defoliation. This was done by building annualized, species-specific modifiers for the key increment and mortality equations using available permanent plots in the region. For commercial thinning, the modifiers are driven by the time since thinning, the intensity of thinning as assessed by proportion of basal area removed, and the type of thinning,

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Table 1—Key base equations and modifiers in the Acadian Variant of the Forest Vegetation Simulator (FVS-ACD) with references

Equation	Reference
Base equations	
Climate site index	Weiskittel and others (2010); Jiang and others (2014)
Crown width	Russell and Weiskittel (2011)
Total height	Rijal and others (2012b)
Height to crown base	Rijal and others (2012a)
Diameter increment	Weiskittel and others (2012)
Height increment	Russell and others (2014)
Height to crown base increment	Russell and others (2014)
Mortality	Weiskittel and others (2012)
Stem taper	Li and others (2012); Weiskittel and Li (2012)
Bark thickness	Li and Weiskittel (2011); Weiskittel and Li (2012)
Ingrowth	Li and others (2011)
Modifier equations	
Commercial thinning (spruce-fir)	Kuehne and others (2016)
Spruce budworm (spruce-fir)	Cen and others (2017)

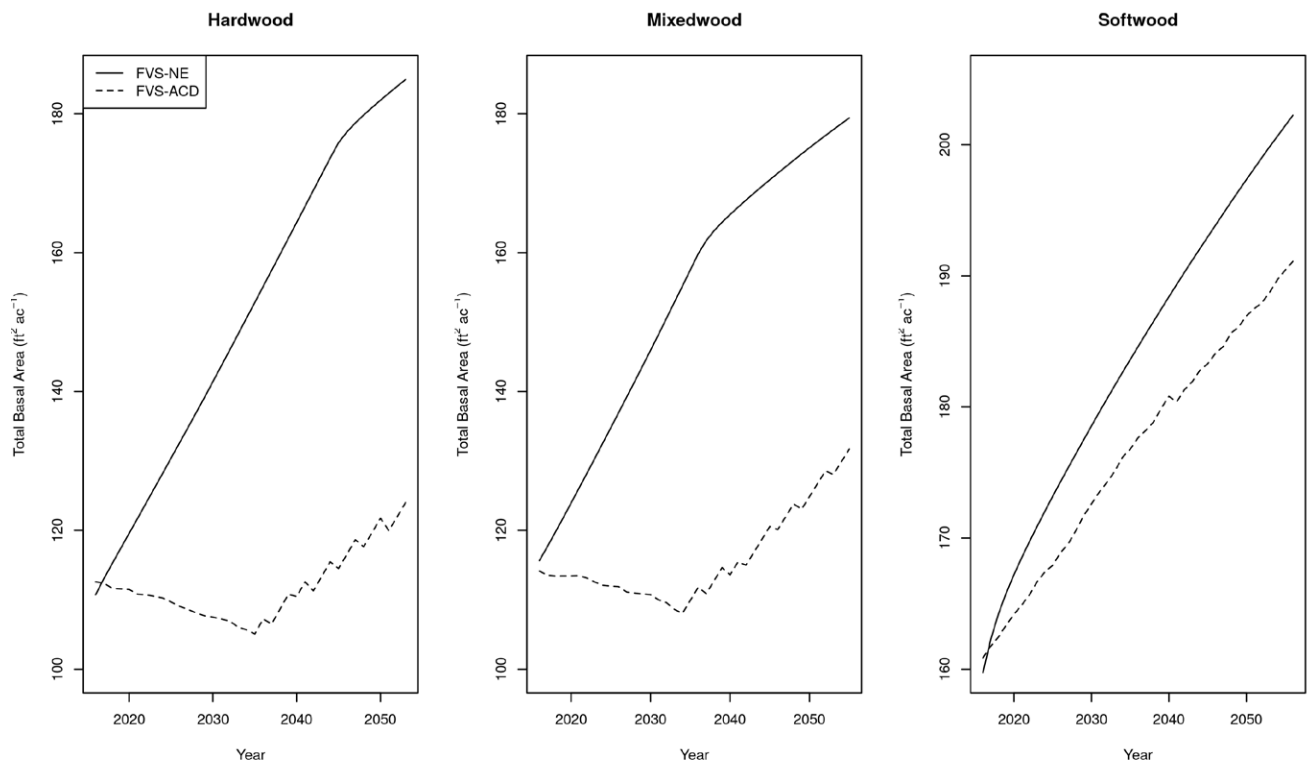


Figure 1—Comparison of Northeast (FVS-NE) and Acadian (FVS-ACD) 50-year predictions of total basal area (ft² ac⁻¹) for three common forest types in Maine.

which is quantified by the ratio of the quadratic mean diameter before and after thinning (Kuehne and others 2016). As expected, the modifiers show increased diameter increment as well as reduced crown recession and mortality following commercial thinning, particularly in heavy crown thinnings. The response generally peaks 4-6 years following thinning and then returns back to normal levels 10-20 years post-thinning. Future efforts will develop similar modifiers for commercial hardwood species. For spruce budworm, the modifiers were a function of various stand structure and composition factors as well as cumulative defoliation expressed as percentage (Chen and others 2017). The modifiers significantly reduce growth and increase mortality, particularly for balsam fir [*Abies balsamea* (L.) Mill.] and cumulative defoliation values above 100-200 percent. These modifiers are applied for the duration of a spruce budworm outbreak, which generally last 10-20 years. Consequently, they can have a significant influence on long-term projections.

Recently, two additional functions were included to better constrain predictions. First, relative density was previously calculated using the maximum size density index (SDI) equation provided in Woodall and others (2005). However, this was found to give maximum SDIs that were too high and low for stands with lower and higher specific gravities, respectively. A new maximum SDI equation was refitted using data from the Acadian with an intercept and slope much lower than those provided in Woodall and others (2005). The final equation was $1830.1 - 1638.4 * SG$ where SG is the average specific gravity in the stand. Second, total tree heights were constrained by reducing predicted height increment when a tree reached 80 percent of the 99th percentile height observed for that species in the regional database. Both constraints ensured reasonable values and model behavior even after 100-year projections.

Model improvements and enhancements will continue into the future. Focus will now shift from commercial softwoods to commercial hardwood due to a growing presence of those species in Maine and increased interest due to recent shifts in product markets. These enhancements will include the thinning modifiers previously mentioned as well as incorporation of equations that assign trees into

alternative hardwood stem form (ideal, acceptable, poor) and risk (low, moderate, high), which will influence tree growth, mortality, and potential product recovery. These equations will primarily be based on the recent research findings of Castle (2017) as well as Castle and others (2017). All of the equations have been fully incorporated as an option under the NE Variant in FVS-Online (<http://forest.moscowfsl.wsu.edu:3838/FVSONline/>), which will also continue to be maintained and updated into the future.

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Development and Evaluation of an Individual Tree Growth and Yield Model for the Adirondacks Region of New York

Aaron Weiskittel, Christian Kuehne, John Paul McTague, and Mike Oppenheimer¹

The Adirondacks Region of New York is characterized by a unique mountain range with climate and soil conditions that vary dramatically with elevation. The forests in the Adirondacks Region of New York are a complex mixture of hardwood and softwood species that have a long and varied history of natural disturbance occurrences and human management. The region is considered an ecotone at the southernmost end of the eastern forest-boreal ecoregion with over 25 different tree species present. A relatively limited number of growth and yield simulators exist for the Adirondacks Region and recent work has suggested these to have some important limitations in the region. The goal of this project was to develop an individual-tree growth and yield simulator that is specific to the Adirondacks Region of New York. Specific objectives were to: (1) test the component equations of Northeast variant of the Forest Vegetation Simulator (NE) for bias in the Adirondacks Region; (2) refit component equations; and (3) evaluate and present long-term prediction behavior.

The data used in this analysis was obtained from long-term continuous forest inventory (CFI) plots located in the privately owned Shirley Forest and four experimental forests maintained and managed by the State University of New York College of Environmental Science and Forestry. Details are provided in Weiskittel and others (2016). All available measurements from the five studied forests not taken after cleaning or harvesting operations were standardized, merged into a common format, and converted to metric units. This resulted in a dataset with 45,496 observations

with 16.6 percent and 15.7 percent having total and bole height measurements, respectively (Weiskittel and others 2016). To evaluate the suitability of NE component equations, an equivalence test with 15 percent allowable error was conducted. Equations were then fit to the Adirondacks data and to the primary species using the programming software R and nonlinear mixed effects modeling (NLME). The primary species included in this analysis were American beech (*Fagus grandifolia* Ehrh.), ashes (white (*Fraxinus americana* L.) and black ash (*F. nigra* Marshall)), black cherry (*Prunus serotina* Ehrh.), balsam fir (*Abies balsamea* L.), eastern hemlock (*Tsuga canadensis* (L.) Carr.), quaking aspen (*Populus tremuloides* Michx.), red maple (*Acer rubrum* L.), red pine (*Pinus resinosa* Ait.), red oak (*Quercus rubra* L.), sugar maple (*Acer saccharum* Marsh.), spruces (black (*Picea mariana* (Miller) B.S.P), red (*P. rubens* Sarg.), and white spruce (*P. glauca* (Moench) Voss.), northern white cedar (*Thuja occidentalis* L.), white pine (*Pinus strobus* L.), yellow birch (*Betula alleghaniensis* Britton), other hardwood species, and other softwood species.

A modified Chapman-Richards equation form was used for the prediction and imputation of total tree height (HT, m):

$$HT = b_{10} \cdot (1 - \exp(-b_{11} \cdot DBH))^{(b_{12} + b_{13} \cdot \ln(BA+1) + b_{14} \cdot \ln(BAL+1))} \quad (1)$$

where

DBH is diameter at breast height (cm), BA is total basal area ($m^2 \text{ ha}^{-1}$), and BAL is the basal area in

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larger trees ($\text{m}^2 \text{ha}^{-1}$). A modified logistic equation was used for the prediction and imputation of bole height (BHT—distance between a 30 cm stump and either a 10 cm upper–stem diameter outside bark or where the central stem terminates due to forking, m):

$$\text{BHT} = \frac{\text{HT}}{\left(1 + \exp\left(\frac{b_{20} + b_{21} \cdot \text{HT} + b_{22} \cdot \ln(\text{BAL} + 0.1) + b_{23} \cdot \ln(\text{BA}) + b_{24} \cdot \ln\left(\frac{\text{DBH}}{\text{HT}}\right) + b_{25} \cdot \text{CSI}}{b_{24} \cdot \ln\left(\frac{\text{DBH}}{\text{HT}}\right) + b_{25} \cdot \text{CSI}}\right)\right)} \quad (2)$$

where

CSI is climate site index (m, Weiskittel and others 2011), and all other variables are defined previously. Individual tree diameter increment was modeled as follows:

$$\Delta\text{DBH} = \exp\left(\frac{b_{30} + b_{31} \cdot \ln(\text{DBH}) + b_{32} \cdot \text{DBH} + b_{33} \cdot (\text{BAL}) + b_{34} \cdot \sqrt{\text{BA}} + b_{35} \cdot \text{CSI}}{b_{34} \cdot \sqrt{\text{BA}} + b_{35} \cdot \text{CSI}}\right) \quad (3)$$

where

ΔDBH is the annual diameter increment (cm yr^{-1}) and all other variables have been defined above. The following height increment model was shown to perform best:

$$\Delta\text{HT} = \exp\left(\frac{b_{40} + b_{41} \cdot \text{HT} + b_{42} \cdot \ln(\text{HT}) + b_{43} \cdot \left(\frac{\text{HT}}{\text{DBH}}\right) + b_{44} \cdot (\text{BAL}) + b_{45} \cdot \text{BA} + b_{46} \cdot \ln(\text{BA})}{b_{44} \cdot (\text{BAL}) + b_{45} \cdot \text{BA} + b_{46} \cdot \ln(\text{BA})}\right) \quad (4)$$

where

ΔHT is the annual height increment (m yr^{-1}) and all other variables have been defined previously.

The model was fit by treating b_{40} and b_{41} as random parameters that varied by species. A logistic function was used to model the probability of individual tree survival:

(5)

$$\text{PS} = \left(1 + \exp\left(-\left(\frac{b_{50} + b_{51} \cdot \text{DBH} + b_{52} \cdot \ln(\text{DBH}) + b_{53} \cdot \text{CR} + b_{54} \cdot \text{BAL} + b_{55} \cdot \sqrt{\text{BA}} + b_{56} \cdot \text{CSI}}{b_{55} \cdot \sqrt{\text{BA}} + b_{56} \cdot \text{CSI}}\right)\right)\right)^{-1}$$

where

PS is the probability of annual survival and all other variables have been defined previously.

Since annual parameters were desired, parameters for the ΔDBH , ΔHT , and PS equations were annualized using an iterative technique. To evaluate the long-term behavior of the equations, a simulation model was constructed by linking all of the component equations. Due to its importance on long-term simulations, the prediction of tree mortality was handled in two ways, namely an expansion factor method (a tree's expansion factor was annually multiplied by the probability of survival) and a fixed cutpoint (optimal cutpoint derived from the species specific survival equations).

For most species and tree attributes, equivalence tests suggested that the observed values and the predicted values from NE were statistically different. The Adirondacks Region thus appears to be a distinct ecological area that is deserving of a growth model specific to the present conditions. In general, the derived component equations fit well and showed adequate performance when conducting long-term simulations. Consistent with other tree-level growth models, total height equations fit the best, while height increment and mortality equations were the most problematic. The diameter and height increment equations proved particularly challenging due to remeasurement data only being available for trees >10 cm in DBH. However, relatively well-behaved and logical increment equations were constructed. For mortality, the area under the curve (AUC) for most species was approximately 0.75, which represents an acceptable to excellent discrimination of alive and dead trees.

When the equations were combined into a growth and yield system, long-term behavior was consistent with observed trends and general expectations despite high variability in the data and incomplete histories of past stand disturbances and

harvesting practices. Interestingly, long-term model performance was improved when using a whole-tree rather than expansion factor approach to individual tree survival (fig. 1). Long-term prediction accuracy of the derived growth and yield system was

slightly better when compared to NE (table 1). The equations presented here have been integrated into FVSONline (Crookston and Shettles 2017) through the AdirondackGY run script option for stands running the NE variant.

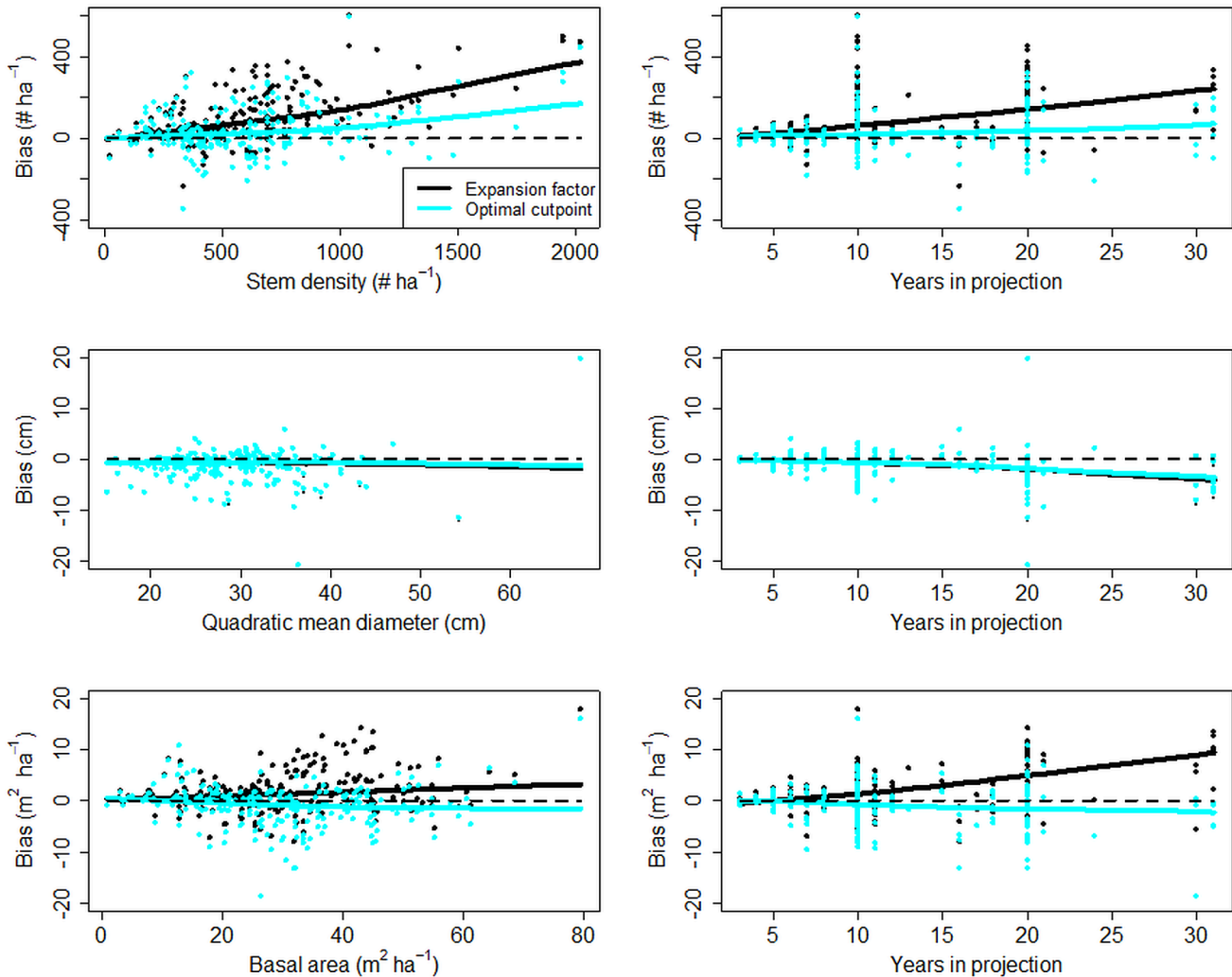


Figure 1—Prediction bias (observed – predicted) for stem density (stems ha⁻¹), quadratic mean diameter (cm), and total basal area (m² ha⁻¹) over observed values (left) and years in projection (right) using the different methods for simulating individual tree mortality. The expansion factor method is where the predicted probability of survival is multiplied by the tree’s current expansion factor, while the optimal cutpoint method is where an entire tree record is killed when the predicted probability of survival falls below the species optimal cutpoint.

Table 1—Stand-level projection mean bias (MB; observed – predicted) and root mean square error (RMSE) for total stem density (stems ha⁻¹), quadratic mean diameter (QMD; cm), and basal area (m² ha⁻¹) by growth and yield system and forest

Forest	N	Years simulated (min – max)	Stem density (stems ha ⁻¹)		QMD (cm)		Basal area (m ² ha ⁻¹)	
			MB	RMSE	MB	RMSE	MB	RMSE
This study (Optimal Cutpoint method)								
DMF	10	8.9 (7 – 17)	-18.53	77.85	-0.28	1.46	-1.54	3.93
HWF	59	20.1 (10 – 31)	74.97	136.73	-2.97	5.67	-1.09	5.28
PDF	71	10.0 (10 – 10)	58.82	143.28	-0.96	1.83	-1.39	4.09
PEF	2	8.5 (7 – 10)	49.42	55.25	-1.30	1.47	-0.19	0.35
SF	103	8.7 (3 – 24)	-5.64	53.52	-0.08	1.55	-0.51	2.83
Overall	245	11.8 (3 – 31)	32.38	109.22	-1.05	3.14	-0.95	3.94
FVS-NE								
DMF	10	8.9 (7 – 17)	3.95	74.22	1.28	2.31	-0.43	3.43
HWF	59	20.1 (10 – 31)	104.83	153.10	3.25	7.05	1.75	5.53
PDF	71	10.0 (10 – 10)	99.89	175.86	-1.77	2.58	-0.21	4.15
PEF	2	8.5 (7 – 10)	58.07	62.64	2.64	2.87	0.46	0.51
SF	103	8.7 (3 – 24)	-33.78	127.46	0.34	1.59	-1.68	7.51
Overall	245	11.8 (3 – 31)	40.63	147.29	0.49	3.90	-0.36	6.05

DMF = Dubuar Memorial Forest; HWF = Huntington Wildlife Forest; PDF = Pack Demonstration Forest; PEF = Pack Experimental Forest; SF = Shirley Forest; MB = mean bias; RMSE = root mean square error; QMD = quadratic mean diameter.

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Modeling Future Dynamics of European Beech Forests for Ground Beetle Conservation

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Carabus olympiae is a steno-endemic ground beetle that has its elective habitat in alpine shrubland and beech (*Fagus sylvatica*) forests of the Valle Sessera (Italy, 45°40' N, 8°16' E) (Negro and others 2008). Coppice with standards proved a more favorable habitat than even-aged high forests, due to its higher structural heterogeneity (Negro and others 2014). However, it is unclear whether the recent abandonment of coppicing will improve or deteriorate the habitat for this and other carabid species (Negro and others 2013), and what management actions are needed to preserve it. We used habitat modeling and stand projection by the Forest Vegetation Simulator (FVS) to understand the response of carabid habitat to future forest dynamics and management.

We measured tree characteristics and growth in 30 beech coppice stands, of which 16 had been abandoned for 50 years, and 14 converted to high forest by repeated thinning starting in 1980. We then calculated the following stand-scale descriptors of forest structure, and compared them between coppice and high forest stands by Wilcoxon test: trees per hectare (TPHA), quadratic mean diameter (QMD), basal area (BA), relative BA of beech, standard deviation of stem diameter (stDBH), volume of coarse woody debris (CWD), and percent canopy cover (CC) assessed by hemispherical photographs. In each stand, ground beetles were caught by means of pitfall trapping from May to August 2013. We fitted generalized linear models of *C. olympiae* abundance, total ground beetle abundance, and relative Shannon diversity as a function of all forest structure descriptors, slope, and aspect. Models were pruned by backward stepwise selection.

FVS was initialized with measured tree- and stand-level data, and calibrated by keywords (Castaldi and others 2016, Vacchiano and others 2014). Site index and maximum Stand Density Index (SDI) were calculated from field measurements. A new equation for crown width and a multiplier for crown ratios were fitted using field measurement data, while height-diameter and large tree growth submodels were allowed to self-adjust based on measurements. We used the FVSONline Northeast variant of FVS (NE) with the Fire and Fuel Extension, due to its simple diameter growth submodel and existing parameterisation for *Fagus grandifolia*. Stand dynamics were simulated in all stands for 2012-2112 with no sprouting under five different treatments: (1) control, (2) conversion to high forest by low thinning, (3) conversion by high thinning (target SDI = 210), (4) single tree selection (30 percent largest trees removed), and (5) group selection with natural regeneration (600 trees per ha) every 20 years. Thinning and single tree selection were triggered if relative SDI > 0.6. Finally, we calculated the expected habitat metrics for ground beetles by extrapolating their respective models over the simulated stand characteristics.

Abundance and diversity of ground beetles was positively influenced by QMD, TPHA, CC, CWD, and stDBH, and negatively influenced by slope and basal area (deviance explained = 36-56 percent) (table 1). Total abundance was mostly influenced by stDBH and Shannon diversity by QMD. Coppices and high forests differed significantly (mean QMD: 12 vs. 20 cm, mean TPHA: 2200 vs. 850) even under similar basal area and canopy cover. Without management, all stands exhibited similar end-of-rotation basal area (50-55 m²), CWD volume

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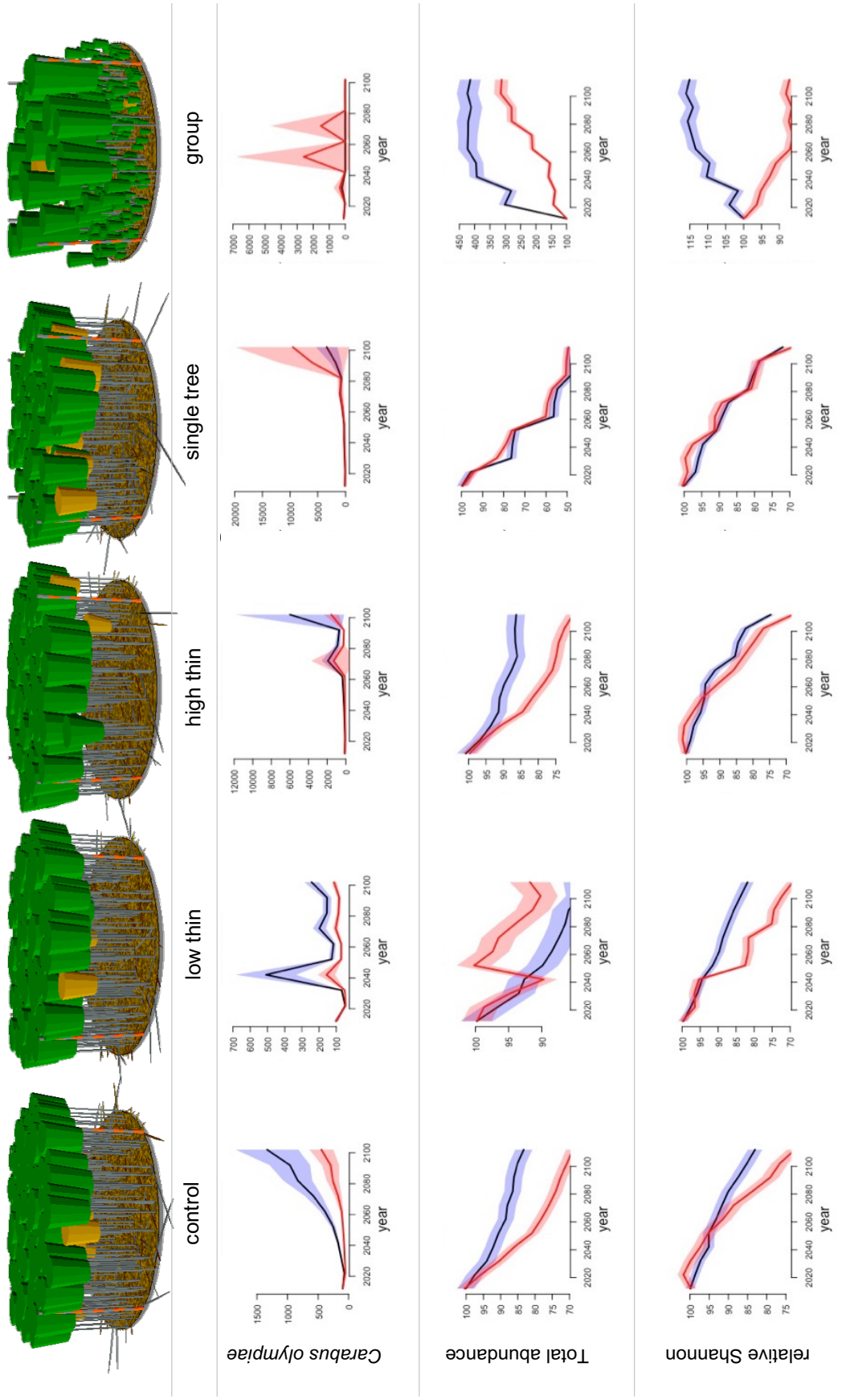


Figure 1 – Simulation of ground beetle habitat metrics under alternative management choices for abandoned coppices (blue) and high forests (red).

Table 1—Generalized linear models of ground beetle abundance and diversity (after backward stepwise selection based on Akaike’s Information Criterion).

<i>Carabus olympiae</i> abundance		
	b	sig.
BA	-0.053	–
CC	0.059	**
TPHA	0.001	**
QMD	0.111	*
CWD	0.027	–
Slope	-0.024	**
Deviance explained: 43 percent		
Total abundance		
sdDBH	0.209	***
TPHA	0.000	–
Slope	-0.014	*
Deviance explained: 36 percent		
Shannon index		
BA	-0.002	*
QMD	0.007	***
Slope	0.002	***
Deviance explained: 56 percent		

BA = basal area; CC = canopy cover; TPHA = trees per hectare; QMD = quadratic mean diameter; CWD = coarse woody debris; STDBH = standard deviation of stem diameter.

(90–110 m³ ha⁻¹), and mortality (self-thinning to around 1000 TPHA), with coppices showing a higher BA (+10 percent) and lower CWD (-15 percent) than high forests after 100 years. Low thinning increased coppice stDBH and reduced CWD relative to the control; high thinning and single tree selection reduced high-forest BA and increased CWD; group selection reduced BA (-50 percent) and CWD, and greatly increased size heterogeneity, especially in high forests. Habitat metrics responded by an increased abundance of *C. olympiae* in unmanaged and high-thinned coppices, and in high forests only under single tree or group selection (fig. 1). Carabid abundance and diversity were higher in coppices, but always declined throughout the simulation, except in low-thinned high forests and under group selection.

Modeling allowed us to understand the effect of several interacting forest variables on ground beetle habitat, and to compare the consequences of management choices. Abandoned coppices will still be suitable for *C. olympiae*; high forests should be subject to single tree or group selection rather than the traditional low thinning. Group selection is the best option to maintain or improve habitat for *C. olympiae*, and abundance and diversity of all ground beetles, which would otherwise develop in contrasting directions (Toïgo and others 2013). High levels of size heterogeneity and deadwood create a better habitat mosaic for all beetles and their preys, and are compatible with variable retention that targets economic return by improving beech stem quality. Future research will target the short-term response of ground beetles to harvest, and provide guidelines for less impacting logging.

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Development of Extensions, Post Processors, and Links to Other Models

The FVS-WRENSS Water Yield Post-Processor: Validation of Snow-Dominated Procedures

Robert N. Havis¹

ABSTRACT—Forests provide about two thirds of the nation’s freshwater with about half originating on federal forests and grasslands in the West. To assist forest managers in optimizing the delivery of freshwater supplies from forested land, a water yield post-processor for the Forest Vegetation Simulator (FVS) model, a widely used forest management tool, has been developed for the contiguous United States. Validation of the FVS-WRENSS water yield post-processor used data from a harvesting experiment in the Fool Creek watershed at the Fraser Experimental Forest in the Colorado Rocky Mountains, where one half of the forested land was cut-block harvested in a paired watershed study. FVS was initialized with stand inventory data and used to simulate undisturbed forest growth, the historic harvesting, and the subsequent forest regeneration and regrowth. Minor adjustment of the meteorological input data to FVS-WRENSS was required to accurately simulate the magnitude and trends in water yield change over 21 years. Simulated average stream flow was within 10 percent of field measurements.

INTRODUCTION

Forests cover about one-third of the United States and are the source of one half to two-thirds of the nation’s high quality freshwater supply (Brown and others 2008, Chang 2012, Shifley 2012). Approximately 18 percent of the water supply originates from National forests and grasslands nationwide. In the West, 51 percent of the water supply originates from National forests and grasslands (Brown and others 2008). With increased population and industrial pressures on forests, and the uncertainty of changing climate impacts on forest health, the protection of water resources is an important objective in forest planning today. The U.S. National Forest System was created, in part, to protect clean water in forest headwaters through the Forest Organic Act of 1897. The Weeks Act of 1911 allowed the use of public funds to purchase land to protect navigable waterways and headwaters in the Eastern United States. The National Forest System is mandated (National Forest Management Act, NFMA, of 1976, P.L. 94-588) to maintain forest plans, and the U.S. Forest Service’s 2012 Planning Rule requires consideration of ecosystem services as part of integrated resource management. Additionally, a White House Council on Environmental Quality released a policy memorandum in the fall of 2015 directing Federal

agencies to incorporate ecosystem services into Federal decisionmaking.

Most National forests, and many projects on federal, State, and private lands use the Forest Vegetation Simulator (FVS) model (Dixon 2015) in developing their vegetation management plans. In roughly 45 years of research and development the FVS framework has grown in ecosystem services scope to encompass the simulation of wildlife habitat (Crookston and Dixon 2005), evaluating forest carbon sequestration (Hoover and Rebaun 2014, MacLean and others 2014), fire and fuels management (Noonan-Wright and others 2014) and forest management under climate change (Crookston and others 2007). The Water Resources Evaluation of Non-point Silvicultural Sources (WRENSS) (Troendle and Leaf 1980) Handbook procedures are an appropriate system to add to the FVS framework. The procedures have been adapted to use data from FVS simulations (basal area, tree height, stand area, aspect, and elevation) to estimate the effects of silvicultural treatments and disturbance on forest water yield. The benefits of optimizing water yield during planning has economic and social value and the additional effort to run the FVS-WRENSS water yield analyses while performing FVS forest growth projections is minimal.

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Both FVS and WRENSS operate on similar time and spatial scales, and like FVS the WRENSS model has been calibrated to data throughout most of the United States (fig. 1A). The addition of water yield analysis into the FVS model framework creates an efficient system for optimizing the delivery of fresh water supplies from forested lands as part of ecosystem services studies (Sánchez Meador and others 2015). This paper demonstrates the effectiveness of the FVS-WRENSS post-processor by using FVS to simulate a timber harvest experiment between 1954 and 1956 at the Fraser Experimental Forest (FEF) in western Colorado, and using the FVS-WRENSS post-processor to predict the change in water yield caused by the harvest. The effects of simulated harvesting and regrowth on predicted changes in water yield are consistent with measured data from the paired watershed study.

The increase in water yield with reduction in cover and the decrease in yield with increase in cover is well documented (Bosch and Hewlett 1982, Ice and Stednick 2004, MacDonald and Stednick 2003, Stednick 1996) but predicting these trends is complex (Bosch and Hewlett 1982). The largest increases in yield are achieved under the highest precipitation rates. Although species mix is important, in the semi-arid climate of Colorado little change in yield occurs when the average annual precipitation is < 18-19 inches (46-48 cm) or the reduction in cover is < 15 percent (MacDonald and Stednick 2003). In the Central Plains, detectable changes in water yield may require harvesting 50 percent of the forest basal area (Stednick 1996). As a general rule, cover reductions of < 20 percent cannot be statistically detected as streamflow (Bosch and Hewlett 1982).

The Forest Vegetation Simulator Model

The Forest Vegetation Simulator (FVS) model is a suite of computer software designed to simulate forest growth and yield under alternative management scenarios. Inventory data can be automatically translated from corporate databases into FVS input format. These national databases include the U.S Forest Service (USFS) FSveg and Forest Inventory and Analysis (FIA) databases, and the Bureau of Indian Affairs (BIA) InfoDat and the Bureau of Land Management (BLM) EcoSurvey databases. FVS was first developed for northern

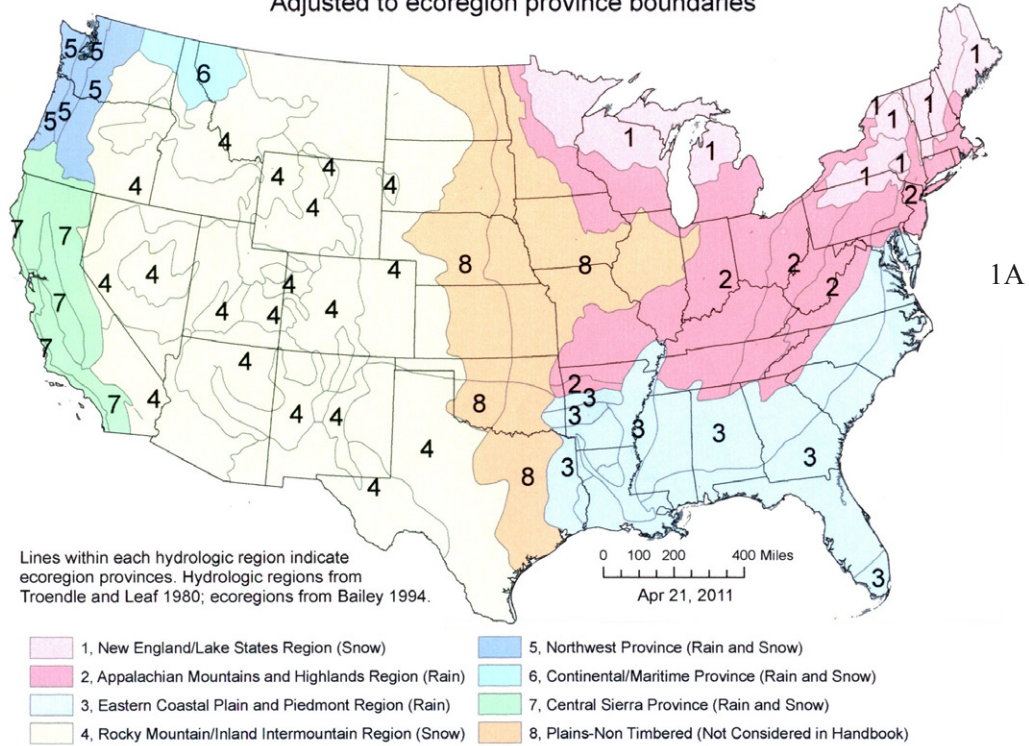
Idaho (Stage 1973) in the 1970s and has been fit to most of the forested areas of the continental United States (fig. 1B). It is an individual tree, distance-independent model that simulates forest growth and mortality dynamics on a stand level. Tree growth functions are often, depending on the geographic variant, fit on the National Forest level, and the model is self-calibrating when local growth information is entered in the input data. FVS considers tree competition and stand density in growth and mortality calculations as well as forest disturbance such as insect and disease, fire, and blowdown. Almost any imaginable harvesting, thinning, pruning, or fertilization management action can be simulated. Stand interactions are not simulated but many thousands of stands can be processed in a single run making it easily scalable to entire watersheds.

The FVS-WRENSS Water Yield Post-Processor

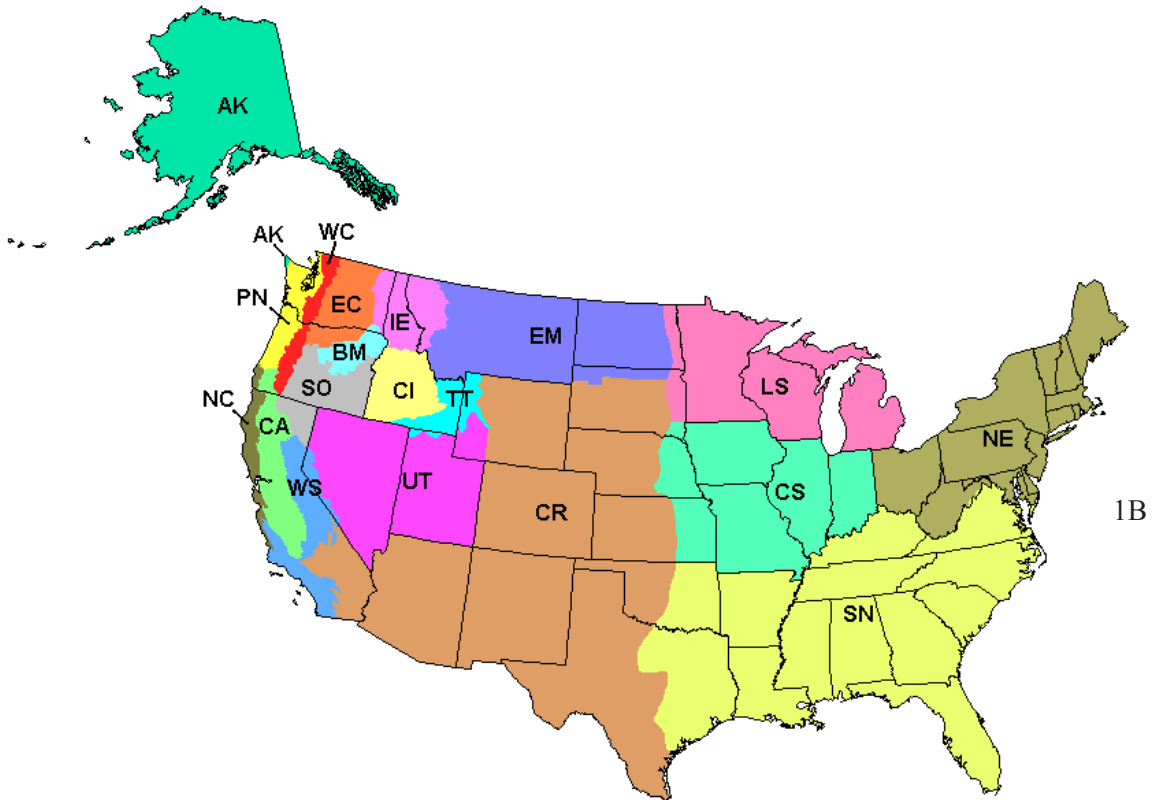
The WRENSS model is documented in chapter 3 of the WRENSS Handbook (Troendle and Leaf 1980). A tabular system is used to perform the water yield calculations. Automated WRENSS computations were programmed by Bernier (1986) for the snow-dominated areas of hydrologic provinces 1, 4, 5, and 6. Huff and others (1999) expanded Bernier's Fortran program to both rain- and snow-dominated procedures in the Central Sierra region of the United States and applied the program to a GIS analysis of water yield changes (Huff and others 2002). Swanson (2004, WinWrnsHyd User's Manual, unpublished manuscript) programmed both rain- and snow-dominated hydrologic procedures for most of the United States and Canada within an Access database platform. The Fortran version of WRENSS has been extended, using the relationships in the WRENSS Handbook, to cover the rain- and snow-dominated hydrologic provinces of the contiguous United States. An interface to the FVS-WRENSS post-processor allows the user to enter parameters such as rainfall lapse rate, daily snowfall, wind speed, soil rooting depth, number of cut blocks, percent of stand harvested, and the input precipitation file (WRENSS_Guide, 2016).

The WRENSS procedures were developed using both regional empirical relationships and deterministic models. Relationships were developed for regions, or hydrologic provinces, having similar

Hydrologic regions and provinces of the conterminous US Adjusted to ecoregion province boundaries



1A



1B

Figure 1—The FVS-WRENSS postprocessor (fig. 1A, Personal communication 2011. R. Bailey (retired), Fort Collins, CO) and the 20 Forest Vegetation Simulator (FVS) model geographic variants (fig. 1B) have been calibrated to the entire contiguous United States.

dominant forms of precipitation and hydrologic processes. In provinces where snow is the dominant form of precipitation snow interception and snowpack ablation are simulated, whereas in provinces where rainfall is dominant soil rooting depth is important in estimating water yield. The hydrology in some provinces (e.g., provinces 5, 6, and 7; fig. 1A) is driven by snow process at high elevations and rain processes at lower elevations. The original Handbook method for snow redistribution in province 4 has been supplemented by the Modified Rocky Mountain (RM) method which is based on recent research showing that the reduction in effective precipitation is caused by interception losses from the snow covered canopy, and therefore modeled as a linear function of canopy density, rather than redistribution of snow during and after snow events (Troendle and others 2010).

Flow routing is not considered in the WRENSS model, water yield is a lumped variable representing the water available for streamflow on a seasonal or annual basis. WRENSS estimates the only loss evapotranspiration (ET) from the hydrologic system and does not differentiate between stream base flow and overland runoff, so that WRENSS calculates water yield as

Water Yield = Precipitation – Evapotranspiration

ET calculations are based on energy, stand elevation and area, user input precipitation, species composition, and FVS predictions of forest density. Except for the Modified RM procedure, which uses stand basal area directly, WRENSS Handbook procedures for each hydrologic province are used to convert stand basal area, predicted by FVS, to cover density or leaf area index. Forest density is normalized to a percentage by dividing the stand density (basal area, cover density, or leaf area index) by the density representing the point of complete hydrologic utilization for the site which is based on local site conditions and stand species mix. Empirical procedures, specific to each hydrologic province, are used to calculate potential ET, and the ET modifier coefficients which are functions of stand relative density, aspect, elevation, and season.

FVS-WRENSS uses monthly precipitation data in the standard format of State climate normals tables, and site-specific data may be used when available. Users may enter a rainfall lapse rate to adjust the input precipitation rates to account for differences in elevation between the meteorological station and the study site. The monthly precipitation data is summed into seasonal totals for the ET calculations. The seasonal water balance and annual summaries are output using the same time increments as the FVS simulations. The water balance calculations for individual stands are weighted by stand area and summarized for each FVS run. Therefore, if an FVS run comprises an entire watershed, which can be from one to thousands of stands, the effective water yield is automatically calculated for the entire watershed.

The FVS-WRENSS water yield post-processor is not meant to predict absolute flows. Instead it was developed for comparing alternative management scenarios. This study compares the predicted water yield from an undisturbed forest to the predicted water yield from a forest harvest experiment. The difference between the predicted flows from each simulation are compared to the estimated change in flow measured in a paired watershed study at the U.S. Forest Service's FEF.

METHODS

Study Area

The FEF, near Fraser Colorado, is a 36 square mile (9,324 hectares) facility operated by the USDA Forest Service, Rocky Mountain Research Station (Alexander and Watkins 1977). The Fool Creek and adjacent East St. Louis Creek flows were calibrated between 1943 and 1954 to establish a reference for estimating the future changes in Fool Creek flows caused by harvesting. In 1952 a road system was built on the Fool Creek watershed and 50 percent of the forested land was harvested in 1954, 1955 and 1956. The study has been well documented (Alexander and Watkins 1977, Troendle and King 1985, Troendle and Olsen 1993) and flows from Fool Creek and East St. Louis Creek have been monitored from the dates of harvest, until present.

The Fool Creek watershed is 714 acres (289 hectares) and ranges in elevation from 8,800 to 12,804 feet (2,682 – 3,903 m). One third of the

watershed (164 acres, 66 hectares) lies above timberline. Of the 550 acres (223 hectares) of forest, 55 percent is lodgepole pine (*Pinus contorta* var. *latifolia*) and 45 percent is Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*). Between 1954 and 1956 alternate strip clearing of trees \geq to 4 inches (10 cm) d.b.h., removed 3.5 million board-feet (8,260 m³) from 278 acres (112 hectares), including 35 acres (14 hectares) of roadway. The cut strips were perpendicular to the slope and of 4 widths, 1-, 2-, 3-, and 4-chains (20, 40, 60, and 80 meters) wide (Alexander and Watkins 1977).

Initial Stand Conditions

Forest inventory data are not available from the Fool Creek watershed, but data is available from the Lexen Creek watershed in the FEF only 3.5 miles (5.6 km) west of Fool Creek and at about the same elevation and aspect. The forest inventory data at Lexen Creek was collected in 1986 and 1991 in mature spruce-fir and lodgepole pine forest types (Personal communication. 2016. Wayne D. Shepperd, Silviculturist (retired), Rocky Mountain Research Station, 240 West Prospect Road, Fort Collins, CO 80526). The mature forests in the Lexen Creek and Fool Creek watersheds of the FEF change very little over time and this is illustrated by a comparison of the Lexen Creek stand statistics and measurements about 40 years earlier by Wilm and Dunford (1948) at Fool Creek (table 1). The tree count measurements are very similar in the two forest types and the volumes are comparable. The Lexen Creek forest inventory data shows more

volume (14,000-16,000 board feet per acre, 82-93 m³/ha) than the average measured volume in 1948 but it is within the range, 7,600–17,000 board feet per acre (44-99 m³/ha), of the volume estimates at that time. These data support the use of the Lexen Creek forest inventory data to initialize the FVS simulations of Fool Creek forest growth in the mid-1950s.

The maximum stand density of each forest type was estimated to be 5 percent over the inventory stand basal area. A value of 199 square feet per acre (46 m²/ha) was used for the spruce-fir stand and 191 square feet per acre (44 m²/ha) was used for the lodgepole pine stand. These values are consistent with estimates of full hydrologic utilization for spruce-fir and lodgepole pine of 224 and 191 square feet per acre (51 and 44 m²/ha) respectively in the Platt River Basin of Colorado and Wyoming (Troendle and Nankervis 2014).

Precipitation and Weather Data

Precipitation data is available for the years 1976 - 2003 from the FEF Headquarters weather station at an elevation of 8560 feet (2609 m) (Elder 2005). The earlier Headquarters weather station data from 1956 to 1975 have not been published, so data from the nearby city of Winter Park, CO, for the years 1942-2016, (Western Regional Climate Center, 2215 Raggio Parkway, Reno, NV 89512-1095 <http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?co9175>) was used to estimate the monthly precipitation at the FEF Headquarters. The Winter Park station is at an elevation of 9,058 feet (2761 m) and approximately

Table 1—Initial stand conditions measured in 1986 and 1991 on the Lexen Creek watershed and used in FVS simulations compared to stand conditions measured on Fool Creek watershed in 1948 by Wilm and Dunford (1948)

Fool Creek watershed	Lexen Creek watershed	
all trees	Spruce-Fir, 1986	Lodgepole Pine, 1991
	----- 3.5 in (8.9 cm) d.b.h. \leq tpa -----	
300 – 400	264	357
	-----3.5 in (8.9 cm) d.b.h. \leq tpa \leq 9.5 in (24.1 cm) d.b.h.-----	
147	123	174
	----- board-feet per acre (m ³ /ha) 9.5 in (24.1 cm) \leq d.b.h. -----	
12,000 (70)	16,000 (93)	14,000 (82)

d.b.h. = diameter at breast height.

5 miles (8 km) east of the FEF. Linear regressions were developed between the monthly data from the FEF Headquarters and the Winter Park station for the overlapping years (1976 to 2003). The linear regressions for each month were applied to the 1956 to 1975 Winter Park station precipitation data to estimate the missing monthly precipitation data at the FEF Headquarters during those years. The estimated monthly data was combined with the measured data at the FEF Headquarters to provide the input for the FVS-WRENSS simulations. These data are available in the Colorado state normals data file that is part of the FVS software setup package. The average daily snowfall rate (18 mm/day) input variable was estimated by a count of the average days with precipitation per month for the months of October through February (14 days at the FEF) (Norwegian Meteorological Institute and Norwegian Broadcasting Corporation 2007-2015 http://www.yr.no/place/United_States/Colorado/Fraser_Experimental_Forest/statistics.html) and an estimate of 10 inches of precipitation during the winter months (October – February).

FVS Simulations

Two FVS simulations were configured to calculate the effects of harvesting on water yield, a control or no-action alternative and the cut-block harvest alternative. Because FVS outputs are per-acre averages, cut strips and leave strips are not well represented by a simulation representing a single stand. The entire area would appear to have been partially harvested. To address this in the cut-block harvest alternative, one stand was used to represent the cut strips, with 100 percent of the stand harvested, and another stand was used to represent the leave strips, with none of the stand harvested. The area represented by each stand was adjusted to represent the appropriate proportion in the cut block experiment. Hence the simulation representing the harvest alternative had 5 stands, one spruce-fir stand and one lodgepole pine stand that were harvested representing the cut strips, one spruce-fir stand and one lodgepole pine stand that were not harvested representing the leave strips, and one alpine area. The simulation representing the control or no-action alternative had three stands, the spruce-fir stand, the lodgepole pine stand, and the alpine area.

The simulations used the Central Rockies (CR) variant of FVS (version 1778, revision date

4/7/2016) which does not automatically add regeneration to disturbed lands. Therefore, seedlings were added to the cut spruce-fir stand using data from Alexander and Watkins (1977) who present naturally regenerated seedlings/saplings counts for the 4 cut-widths immediately after harvesting and 10 years after harvesting. There was an average of 1,362 (std=464) Engelmann spruce and 3,500 (std=804) subalpine fir seedlings/saplings immediately after harvest and 1,437 (std=605) Engelmann spruce, and 4,100 (std=867) subalpine fir 10 years after harvest. To simulate the regeneration in the cut areas, 1,362 Engelmann spruce and 3,500 subalpine fir were added to the FVS simulation in 1956 and the difference between the counts, immediately after harvest and 10 years later, 75 Engelmann spruce and 600 subalpine fir, were added in 1966. The two simulations, the no-action and harvest scenarios, were processed for 27 years from 1956 to 1983.

FVS-WRENSS Simulations

The FVS-WRENSS simulations used the stand attributes and vegetative information from the FVS simulations and the monthly precipitation data described earlier. It was assumed that one half of the road area was built in the spruce-fir cut area and the other half was built in the lodgepole pine cut area. So 17.5 acres (7.1 ha) was subtracted from the cut areas in each forest type (table 2). It was assumed that there was no ET from the road area. This is a reasonable assumption since WRENSS estimates seasonal or annual water available for stream flow and there is no transpiration from a roadway and evaporation from the compacted roadway soil would be negligible. Therefore the precipitation falling on the roadway was mixed into the predicted water yield from the harvest simulation assuming no losses. Both the Handbook snow redistribution method and the Modified RM method were used to model snow hydrology. The Handbook method assumes that snow is blown off the canopy and deposited in openings and the FVS-WRENSS input data can be configured to perform these calculations.

The lodgepole pine stand was harvested with 88 cut blocks and the spruce-fir stand with 72 cut blocks (Alexander and Watkins 1977), so an average of 80 cut blocks was used in the FVS-WRENSS simulations. Although FVS-WRENSS is not very

Table 2—FVS stand units used to model the Fool Creek watershed alternative analysis

Alternative	Stand forest type	Treatment	Area
			----- acres (hectares) -----
No-action	Spruce-fir	None	248 (100)
	Lodgepole pine	None	302 (122)
	Alpine	None	164 (66)
Harvest	Spruce-fir	cut, d.b.h. \geq 4 inches (10.2 cm)	107 (43)
	Spruce-fir	None	124 (50)
	Lodgepole pine	cut, d.b.h. \geq 4 inches (10.2 cm)	134 (54)
	Lodgepole pine	None	151 (61)
	Alpine	None	164 (66)
	Roads	None	35 (14)

d.b.h. = diameter at breast height.
 Roads not simulated in FVS.

sensitive to these variables, the size of the cuts is used to estimate wind fetch and snow pack ablation. The forested stands are at an east/west aspect and an average of 11,000 feet (3350 m) elevation, and the alpine area is at a north aspect and an average of 12,000 feet (3660 m) elevation.

RESULTS

FVS Simulations

The FVS model was used to simulate the growth of an undisturbed forest on the Fool Creek watershed and the regrowth and regeneration on the cut-block harvest experiment conducted between 1954 and 1956. The predicted trends in stand density, based on stand basal area from 1956 to 1983, are shown in figure 2. The basal area predictions are relatively constant for the undisturbed forests (dashed lines) remaining at about 190 square feet per acre (44 m²/ha) for the spruce-fir stand and about 180 square feet per acre (41 m²/ha) for the lodgepole pine stand. The basal area of the harvested spruce-fir forest type (solid thin line) increases rapidly in the 1960s and early 1970s because of simulated regeneration, and the residual lodgepole pine basal area (solid thick line) increases more gradually. The vegetative data from FVS were used in the FVS-WRENSS post-processor to simulate the water yield from the no-action and harvest management scenarios. The difference in water yield between the

two simulations was used to predict the change in water yield caused by the harvest.

FVS-WRENSS Simulations

The difference in water yield between the no-action and harvest scenarios is plotted with the data estimated from field measurements (Troendle and King 1985) in figure 3. The rainfall lapse rate was adjusted such that the predicted average annual watershed precipitation over the 27-year simulation (29 inches per year, 740 mm/year) matched the midpoint of the average of 28 to 30 inches per year (710-760 mm/year) estimated by Alexander and Watkins (1977). Very fine adjustments to the rainfall lapse rate (0.65 inches per 1,000 feet, 54 mm/1000 m) and average wind speed (13 km/hour) aligned the predicted (triangles, dashed trendline) and estimated (circles, solid trendline) trendlines in figure 3. While the average model input precipitation matched the field estimate, the simulated average annual Fool Creek flow (1956 to 1983) was 8 percent more (13.2 inches per year, 335 mm/year) than the measured Fool Creek flow (12.1 inches/year, 307 mm/year) calculated from measurements by Elder (2006). The results from the Handbook snow redistribution method in FVS-WRENSS, using the same model parameters, are also plotted (Xs, dash-double dot trendline) in figure 3. The magnitude of the difference between the no-action and harvest scenarios is smaller using

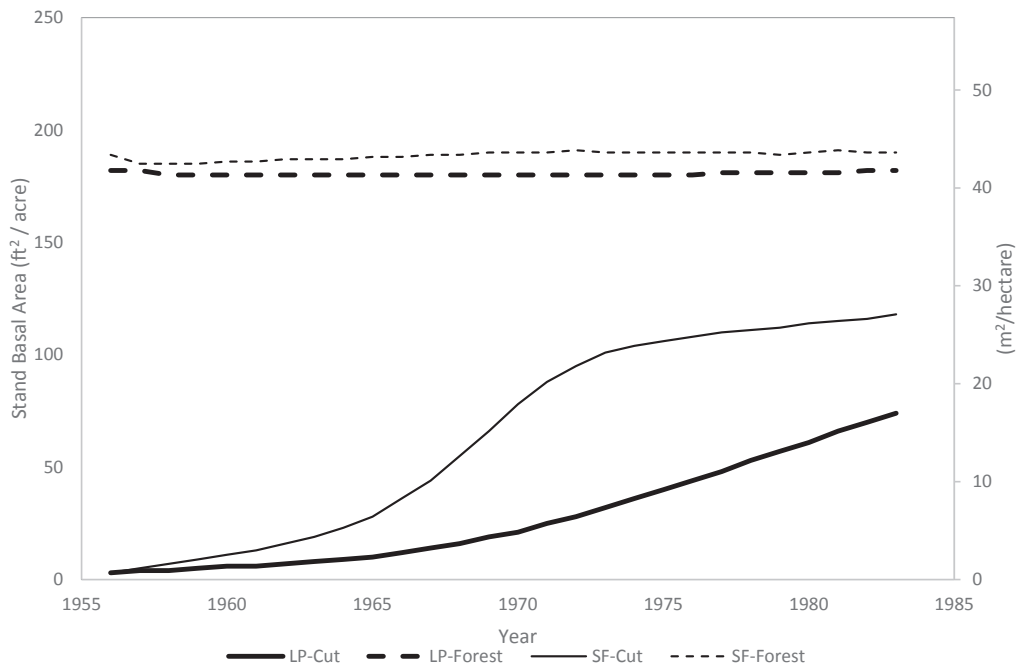


Figure 2—The Forest Vegetation Simulator model (FVS) predicted the growth of the untreated lodgepole pine (LP) and spruce-fir (SF) forests (dashed lines) and the regrowth of the harvested cut-blocks (solid lines). The simulated forest density is shown in terms of stand basal area (ft²/acre).

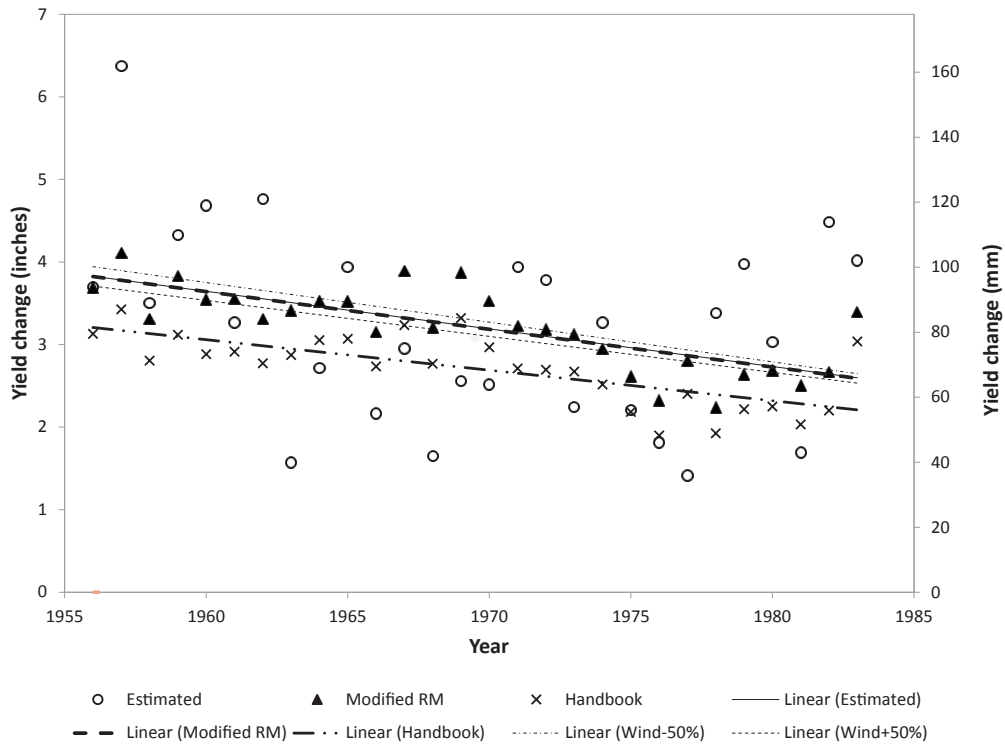


Figure 3—The change in water yield (inches) was estimated using field data (circles, solid trendline) and calculated using the FVS-WRENSS Modified RM snow method (triangles, dashed trendline) and the Handbook snow method (Xs, dash-double dot trendline). The Modified RM trendline is aligned directly over the trendline calculated from the estimated field data. The sensitivity of the trend in water yield recovery (fine dashed lines) was calculated by varying the winter wind speed by 50 percent.

the Handbook snow method than the Modified RM snow method, but the simulated Fool Creek streamflow was 32 percent greater (15.9 inches per year, 404 mm/year) than the measurements by Elder (2006).

DISCUSSION

The FVS model is a widely used tool in the United States to evaluate the vegetative impacts of alternative management scenarios. The results of these forest management analyses may now be evaluated in terms of water resources using the FVS-WRENSS post-processor making FVS a more robust ecosystem services tool. The simulated management alternatives were complex involving significant differences in elevation over the study area, cut-block harvesting methods, snowpack water balance, and flow from a combination of forest and alpine environments. With minimal adjustments to the rainfall lapse rate and average annual wind speed, FVS-WRENSS accurately predicted changes in water yield in a high elevation, snow-dominated system. As well as accurately predicting changes in water yield, the predicted annual average stream flows were within 10 percent of the field measurements. The variability in the estimated annual change in water yield is greater than predicted in the simulations. This is not surprising since the measurements are subject to the runoff dynamics of temperature driven snowmelt events and the FVS-WRENSS simulations predict water available for streamflow without considering runoff dynamics or flow routing. The Modified RM snow method predicted a slightly higher change in water yield than the Handbook snow redistribution method and the predicted annual streamflow was closer to the streamflow measurements than the Handbook method.

CONCLUSIONS

The FVS model framework and the FVS-WRENSS post-processor provide a tool for evaluating the effects of forest management and disturbance on water yield. FVS-WRENSS should be used in forest planning studies where water resources are important. The automated forest inventory data translation between the major land management (FS, BIA, and BLM) databases and FVS makes FVS a robust tool for local, regional, and landscape

ecosystem services analyses. This paper validated FVS-WRENSS water yield predictions in the snow-dominated high-altitude environment of the Colorado Rocky Mountains. It showed that the interception-based snow method performed marginally better than the snow redistribution method for modeling snow hydrology in disturbed environments. However, given the assumptions used to calculate water yield either snow method could be used to compare management alternatives, although the Modified RM method is simpler to implement. Further validation of FVS-WRENSS should be performed using paired watershed data in the other hydrologic provinces of the United States. The FVS-WRENSS system could be enhanced to model water quality thereby furthering its usefulness in supporting ecosystem services objectives in the management of public lands.

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Linking FVS and TELSA via the API

Donald C.E. Robinson and Sarah J. Beukema¹

The Forest Vegetation Simulator (FVS) Application Program Interface (API) was created to allow developers to use all variants of FVS in conjunction with other simulation tools. As described in the open-fvs wiki (Robinson 2015a), the development environment and source code were modified to create a smaller executable program linked to a set of Dynamically Linked Libraries (DLLs). This architectural change allows FVS to be run either as a “classic” command-line simulation or as an embedded simulation component controlled by other supervisory software. At the time the architecture was changed, the development environment was enhanced so that 32- and 64-bit versions of FVS could be compiled for operating systems running Unix-alike or Windows using a single open-source software building system (cmake; <https://cmake.org/overview/>) that supports multiple operating systems and compiling tools. With cmake, a Unix-alike FVS executable and shared object libraries are built using standard Unix software (Robinson 2015b), while Windows executables and DLLs are built with either MinGW (Robinson 2015c) software or Visual Studio 2010 with Intel Fortran (Robinson 2015d). Outputs from FVS builds made with cmake are the same across platforms as well as being identical to outputs from FVS executables (which do not make use of DLLs) released by the U.S. Department of Agriculture Forest Service Forest Management Service Center.

To date the API has been incorporated into applications written in VB.Net, R, and Python. Critically, the API adds the ability to repeatedly stop and start FVS (Crookston 2016), allowing other software to take overall supervisory control, possibly modifying FVS internal variables in the process. In this way, dynamic changes can be made on-the-fly, including modifications to the calculation of growth, regeneration and mortality, as

well as harvest and silvicultural scheduling. When used in a landscape context, it is also possible to apply spatially based management to collections of stands or to develop spatially based disturbance models such as multi-stand fire or epidemic insect outbreaks. Previously, spatially explicit insect outbreak models such as the Westwide Pine Beetle Model (Beukema and others 1997) could only be developed through complex custom programming using the Parallel Processing Extension (Crookston and Stage 1991), which is no longer maintained.

Using the API, we linked FVS to TELSA, a state-transition landscape simulation model written in C++ (Kurz and others 2000). In the linked system, TELSA is the supervisory model that provides a treelist and site information to FVS, which then grows the stand in each landscape polygon each year. FVS passes back to TELSA information such as stand volume that can be used by TELSA to schedule harvesting. TELSA also simulates natural disturbances and initiates regeneration and may pass changed treelists back to the FVS program.

The linked TELSA-FVS system places the FVS API behind a Microsoft VB.NET software layer (fig. 1). Although we could have communicated directly with FVS, we chose to develop this intermediate .NET layer, allowing developers to use any .NET language (C++ in the case of TELSA) to communicate with FVS using a common set of Visual Basic methods, including the capability of mixing metric and imperial units (Robinson 2015e). There is no need to master mixed language calling conventions.

As we developed and tested the TELSA-FVS system it was important to carefully consider and define the roles played by the supervisory program

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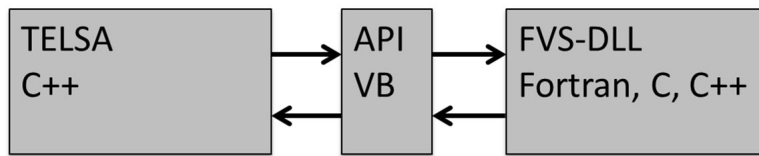


Figure 1—TELSA and FVS communicate through an intermediate .NET layer which simplifies the need to master mixed language programming.

and FVS. This included the following decision points:

- Is TELSAs or FVS used to model mortality?
- Which model is used to model regeneration establishment?
- What tree and site information does the supervisory program need to provide to FVS?
- What FVS output does the supervisory program need to capture?
- What are the timesteps of the supervisory model and FVS model?
- What is the frequency for calling FVS?

Once these roles were clarified we were able to customize TELSAs and link it with FVS, testing landscapes of up to 180,000 stands. We were interested in knowing how runtimes for TELSAs-FVS compared with TELSAs alone, varying the size of the landscape and the simulation complexity. We found that runtimes are controlled by factors that include the size of the landscape, the complexity of the spatial management and disturbance being simulated and file I/O. Linking TELSAs with FVS increases runtime significantly for small landscapes due to the very fast performance of TELSAs on

these landscapes; is about the same for mid-size landscapes and is about twice as costly for very large landscapes due to file I/O (table 1).

When spatial interactions are simulated, FVS is run with the stop and restart capability of the API. In this instance FVS automatically creates two temporary files to hold all model-state information, storing information on the user’s computer when a stand simulation is paused, reloading it when the simulation is restarted. In the process of creating the TELSAs-FVS simulation landscapes we discovered that it was necessary to restrict the number of FVS stands stored in these temporary files, using a batch processing approach to group the FVS simulations managed by TELSAs. In particular, we found that when many stands are stored, the internal index position of FVS variables (an integer) stored within the temporary files can exceed the computer’s ability to store large integers, producing unpredictable outputs and crashes. With further study it may be possible to work around this limit. But as an interim solution we found that creating batches of 5,000 FVS stands, which we implemented in the API, was sufficient to overcome the problem. The upper limit on the number of stands will likely vary with the complexity of the FVS model run (more complex runs store more model-state information), and we recommend that

Table 1—TELSAs-FVS runtimes

Scenario	Stands	TELSAs	TELSAs-FVS	Runtime Ratio
Management only	900	17	31	1.8
	9,000	916	984	1.1
	90,000	738	—	—
Disturbance + Management	900	5	19	3.7
	9,000	401	469	1.2
	90,000	445	930	2.1

Runtime (minutes) for two example landscapes. Blank cells (—) were not simulated.

various batch sizes be tested during application development. Although we did not test different storage media, we expect that intermediate results stored on solid-state storage will execute much faster compared to classic spinning hard drive media.

As a final consideration we note that FVS is limited to 40 simulation timesteps, and this limit may introduce tradeoffs in the temporal extent of the TELSA simulations. For example, if the combined models both use an annual timestep, the overall extent of the simulation will be 40 years. If the supervisory model can be configured to call FVS every 5 years, the overall extent of the simulation may be up to 200 years.

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Linking FVS to 3D Fire Models: Introduction to STANDFIRE, a Platform for Stand-Scale Fuel Treatment Analysis

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Greg Cohn, Rod Linn, Ruddy Mell, and Chad Hoffman¹

With rapid changes in forest health and an increasing presence of fire affecting many landscapes, fuel treatments are considered essential in efforts to potentially mitigate catastrophic fires, restore ecosystems and increase ecosystem resilience. Understanding fuel treatment effectiveness requires quantifying fuel changes and how they translate to changes in fire behavior over time. As these relationships are dynamic and often interrelated in complex ways, modeling-based evaluation efforts, such as with the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) (Rebain and others 2010), play a key role in such analyses. In this paper, we describe STANDFIRE, a prototype platform for modeling wildland fuels and fire behavior at stand scales. STANDFIRE builds upon and extends the capabilities of FFE-FVS by developing 3D fuels inputs for state of the art physics-based fire models, providing a more detailed alternative for analysis of how forest structure and composition may affect fire behavior and effects, particularly with respect to fuel treatment effectiveness. While previous tools simplify fuels data to accommodate fire models, STANDFIRE provides a pathway for researchers and managers in the United States to use real world forest inventory and fuels data in dynamic, 3D fire simulations.

STANDFIRE's modular design connects several components, primarily through text files, facilitating active testing and new science development. Graphical user interfaces are independent and

optional, facilitating batch processing or potential integration with larger systems. STANDFIRE accesses user data through the FVS keyword file and simulates fire for a single representative stand, for a single FVS simulation year, at a time. Multiple runs provide the capability to robustly compare different cases. A simple interface allows the user to browse to the keyword file location, select a year within the FVS simulation, describe field measurable surface fuel characteristics (e.g.; shrub height, fuel load, and percent cover), set the dimensions of the simulation to be carried out, and specify wind speed and ignition conditions. Fire simulations are carried out with a physics-based fire model, the Wildland urban interface Fire Dynamics Simulator (WFDS) (Mell and others 2007). Optionally, input files using the same fuels data may also be built for a different physics-base fire model, FIRETEC, developed at Los Alamos National Laboratory (Linn and others 2005), providing fire researchers with an opportunity for cross-model testing.

STANDFIRE is programmed in python and Java. In python, STANDFIRE uses pyFVS, an open source interface to the FVS library to run the FVS simulation specified in the keyword file. As a 3-D system, STANDFIRE requires spatially explicit data, such as stem mapped stand data. However, because most users do not have such data, as a default, STANDFIRE uses pyFVS to run SVS using the tree coordinates in the 1-acre visualization and statistically extending that forest to a larger area specified by the user (fig. 1). Canopy fuels data,

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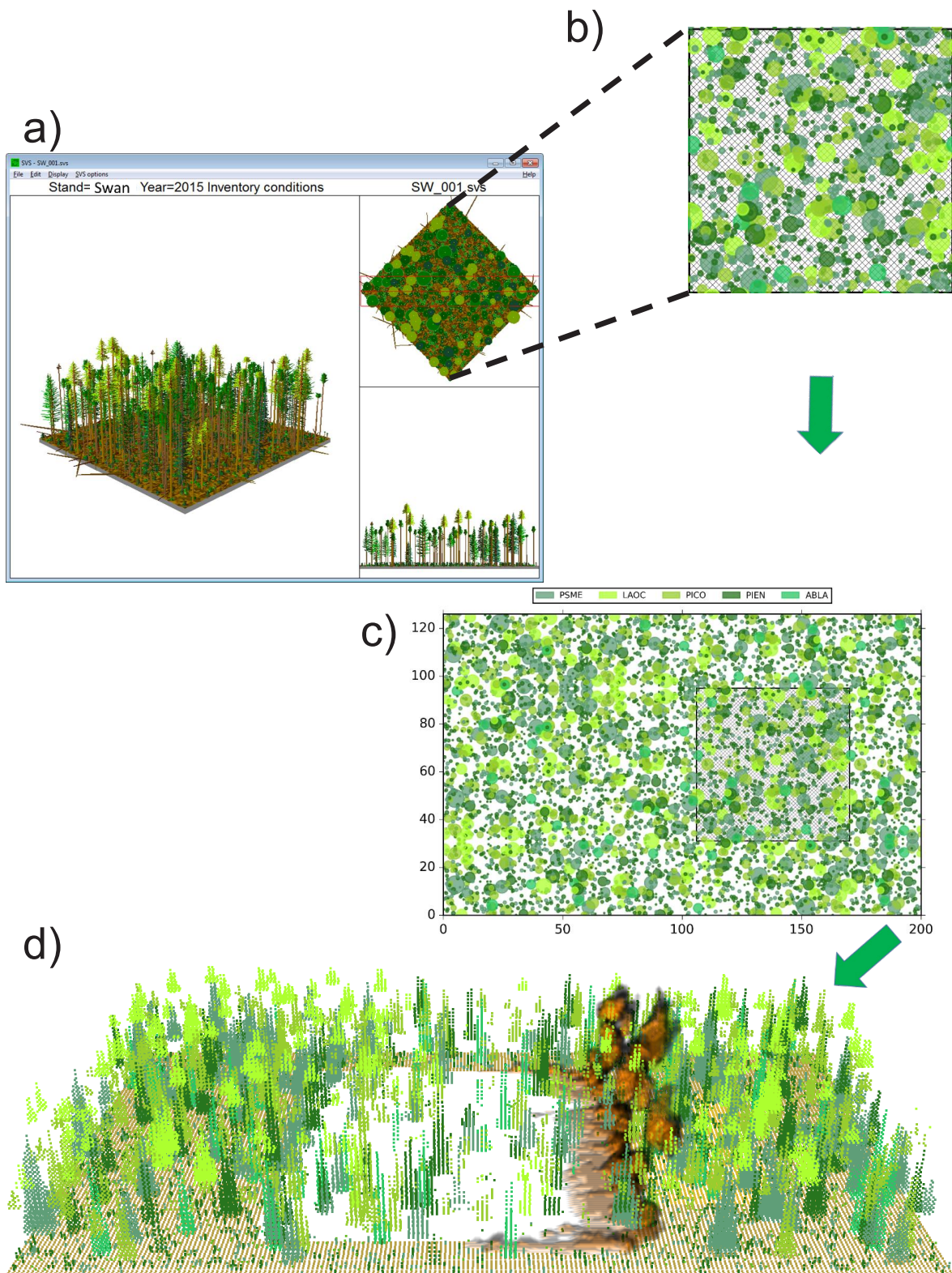


Figure 1—Illustration of STANDFIRE, a prototype system for 3-D fuel and fire modeling at stand scales. STANDFIRE runs FVS and SVS (fig. 1A), and appends biomass data for individual trees from FFE-FVS to the tree coordinates in the one-acre visualization (fig. 1B), statistically extending that forest to a larger area specified by the user (fig. 1C). These data are translated from 2-D to 3-D, populating voxels (3-D cells) with quantitative fuel properties for 3-D fire simulations (fig. 1D).

extracted from FFE-FVS for each tree, are appended to the SVS file, producing a STANDFIRE tree data file. This file, as well as additional files with species and simulation geometry information, are then used by Java-based libraries which implement a state of the art fuel modeling system (Pimont and others 2016) to translate data from 2-D to 3-D, populating voxels (3-D cells) with quantitative fuel properties for 3-D fire simulations. The Java components are built on the Computer Aided Projection of Strategies in Silviculture (CAPSIS) architecture, a collaborative open-source software within which over 60 different forestry-related models have been developed, using a common architecture that provides I/O functions, analysis, visualization tools, shared libraries and source code (Dufour-Kowalski and others 2012). More information on CAPSIS is available at <http://capsis.cirad.fr/capsis/home>.

Fire simulations take several hours and are best carried out on larger areas, with multiple processor machines. Default settings for STANDFIRE, however, will run on less sophisticated computers for small areas. Simulations are highly detailed, accounting for interactions between fuels, fire and wind at fine scales in time and space. STANDFIRE post-processes these complex outputs to summarize a series of metrics quantifying fire behavior, fuel consumption and other aspects characterizing how fire burned through the stand. Development of new metrics is ongoing; an experimental metric links canopy fuel consumption to tree mortality equations, providing spatially explicit, tree level probability of mortality outputs. Other experimental metrics characterize heat release and other fire physics properties.

STANDFIRE is a working prototype system, significant in that it opens the door to new approaches for analyzing how forest changes, either over time, through management activities or other disturbances, affect fire behavior and fire effects. In its current state STANDFIRE will be of use to a broad range of practitioners. As a prototype system, however, it should be considered as a

work in progress. We hope to continue developing and building new capabilities for many years to come. One area in which ongoing work is expected is in continuing validation of physics-based fire models. Although numerous components of these models have been validated in laboratory settings (McDermott and others 2008), field scale validations are challenging, often due to lack of suitable measurements. For this reason, like most models, fire behavior simulation results should be used with caution and with an emphasis on looking at trends (and relative differences) in fire behavior rather than as absolutes. Other future directions include new metrics of fuel and fire behavior changes, LiDAR/stem mapped data inputs, inclusion of topography, and strengthened interactions with FVS and other models. We look forward to collaboration in many of these topics.

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A Framework for Evaluating Forest Restoration Alternatives and their Outcomes, Over Time, to Inform Monitoring: Bioregional Inventory Originated Simulation Under Management

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Abstract—The BioSum modeling framework summarizes current and prospective future forest conditions under alternative management regimes along with their costs, revenues and product yields. BioSum translates Forest Inventory and Analysis (FIA) data for input to the Forest Vegetation Simulator (FVS), summarizes FVS outputs for input to the treatment operations cost model (OpCost) and estimates haul costs for harvested material with the Haul Time model to (1) implement silvicultural sequences; (2) generate harvested tree lists to estimate wood produced and treatment cost; and (3) calculate decadal stand descriptors that characterize management outcomes regarding stand attributes, forest resilience, and carbon dynamics. A BioSum project dataset can support monitoring at Forest and Regional scales by providing initial conditions, and a testbed for evaluating assumptions and potential prescriptions and how their impacts evolve over time. As re-measurements on FIA plots continue over time, they can play a key validation and calibration role, developing new knowledge of management’s latent effects, improvements to future versions of FVS, and refinements in BioSum parameterization. BioSum is a versatile, multi-purpose tool designed to inform managers, planners and decisionmakers charged with sorting through myriad options by highlighting potentially superior choices based on user defined criteria. This paper illustrates the analytic power available via application to the real-world problem of developing fire resilience prescriptions and evaluating the modification in stand trajectories, wildlife habitat related stand attributes, fire resistance, economic trade-offs and logistical considerations that would result from their application in the Western United States.

A BRIEF HISTORY OF BIOSUM

The BioSum framework originated in 2002, when the U.S. Forest Service Pacific Northwest (PNW) Forest Inventory and Analysis (FIA) Program was tasked with estimating how much woody biomass feedstock might feasibly be produced, to supply both wood manufacturing and bioenergy facilities, assuming fuels management was applied over large forested landscapes in southwest Oregon and northern California, Arizona, and New Mexico. We developed a biomass summarization (BioSum) analysis in which we applied the Forest Vegetation Simulator (FVS) as a silvicultural treatment implementation engine to stand data from the many thousands of FIA plots that represented an entire State, or substate region. We relied on the Fire and Fuels Extension to FVS (FFE-FVS) to generate the torching index and crowning index metrics that served as a basis for evaluating and comparing fire

hazard metrics pre- and post-treatment (Fried and others 2005). Treatment costs were estimated with the STHARVEST spreadsheet model (Fight and others 2003), and wood transportation costs using a raster GIS analysis workflow that linked plot locations with existing and proposed processing facilities. There was no projection of stands forward in time, and the FVS database extension did not yet exist. Consequently, FVS text file output had to be parsed with perl and awk scripts and other tools, to fetch desired outputs back to an analysis database where treatment efficacy, wood production and value, and treatment and transportation costs could be summarized and compared (Fried and others 2005). Much of this workflow seems primitive in light of FVS’s current capabilities.

The PNW Research Station’s Focused Science Delivery Program provided significant seed

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funding, generously matched with FIA Program support, to formalize what had been a manual, kludgy, error-prone and problematic hand-cranked “model.” BioSum 3 became a user-friendly tool with workflow management software ready for beta-testing in 2007. The Fuel Reduction Cost Simulator (FRCS) (Fight and others 2006) treatment cost spreadsheet tool was substituted for STHARVEST and a formal spatial analysis workflow was documented to handle haul cost calculation. BioSum 3 was applied to a 25 million acre study area in western Oregon and northern California to demonstrate the proof-of-concept, to characterize the kinds of wood that could be produced by fuel treatments (Barbour and others 2008), and to extend it to include optimization of treatment selection and siting of processing facilities (Daugherty and others 2007).

Analytic capacity was extended in 2011 in BioSum 4 to allow any FIA or calculated FVS data item to participate in the determination of treatment effectiveness. These new capabilities were exercised for the dry mixed conifer fuel synthesis (Jain and others 2012) in which treatment effectiveness was informed by changes to three aspects of fire hazard: (1) fire suppression safety, (2) crown fire severity, and (3) economic impact. These aspects are tied to FFE-FVS predictions of surface flame length, torching index, torching probability, and mortality volume. For the first time, FVS projections were analyzed to understand the carbon implications of fuel treatment under different fire return intervals, considering mortality and harvested products (Fried and others 2013).

BIOSUM 5

The launch of two extramurally funded projects in 2012-2013 made it possible to account for delayed treatment, the possibility of re-treatment, and treatment longevity. BioSum was transformed into a dynamic framework under which many thousands of stands could be treated at multiple time points, and stand attributes under alternative management, including grow-only, could be tracked and compared. Version 5 also brought (1) the introduction of regeneration into BioSum simulations via the REPUTE (Vandendrieche 2010) protocol; (2) the replacement of FRCS with the

OpCost model (Bell and others, 2017a), written in R, developed specifically for use with BioSum; and (3) a computationally fast, graph-theory based haul cost analysis workflow developed with R code in lieu of the previous ArcGIS workflow that was both slow and memory-limited. With these developments, it became clear that BioSum had the potential to be more widely useful, beyond just fuels treatment analyses, for any forest scenario analysis for which it is important to consider broad scale outcomes over a heterogeneous forested landscape. It could be used, for example, to analyze carbon dynamics associated with management and disturbance, considering forest objectives other than fire resilience (e.g., individual or multiple stand attributes related to wildlife habitats), and for analyzing wood supply in a spatially explicit fashion. We are completing a wood supply analysis for BioChar feedstocks as part of a study funded by Oregon State University’s Institute for Working Forest Landscapes. Habitat elements that can be tracked in FVS, such as number of large live and dead trees, canopy cover and down wood, could also be a basis for evaluating the success of silvicultural treatments for achieving desired outcomes under alternative disturbance and climate scenarios.

BioSum 5, renamed “Bioregional Inventory Originated Simulation Under Management” while retaining the existing acronym, marries FIA plot data with the FVS model, and adds custom models for estimating treatment and haul costs, along with a treatment heuristic optimizer. A user can design as many treatments as desired and apply the framework to a landscape as small as a 1 million acre National Forest or as large as the entire Western United States. FIA data has the advantage of informing about both private and public lands—both are needed to truly understand wildlife habitats and other services provided in forested landscapes. Without the BioSum software, work flow management posed a nearly insurmountable challenge given the number of parameters that must be tracked and the large sample sizes that FIA data provide. It is not uncommon for a single BioSum project covering a multi-State area and dozens of management alternatives to grow to over 100GB. It can be helpful to think of BioSum as generating an enormous knowledge base, populated by FVS

output generated via simulating thousands of FIA plots, which comprise a representative sample of the entire forested landscape, using dozens, or even hundreds, of silvicultural prescriptions. In the BioSum simulation environment, FVS's role is to compute relevant stand metrics and apply multiple silvicultural sequences to generate alternative stand trajectories. BioSum is responsible for managing work and data flow, merchandising harvested wood by species and size and moving it to processing facilities. BioSum also estimates treatment cost via OpCost, and supports analysts as they seek to understand the effects and costs of alternative management strategies.

MODEL FRAMEWORK

In essence, BioSum deploys FVS to simulate management of any desired subset of a fully representative sample of all forest based on the consistent, quality controlled field measurements collected by FIA. BioSum also contains a spatial

element to address the location of forests relative to road networks and wood processing facilities, including biorefineries that produce renewable energy. We see it as a potentially valuable tool for management experimentation, because it can generate information about management effects, costs and revenues under alternative objectives, constraints or policies, at much broader spatial scales and in greater levels of complexity than can be achieved using FVS alone. Such pre-implementation knowledge could be thought of as predictive or hypothetical monitoring.

This simplified schematic (fig. 1) traces the workflow beginning with FIA plot data, which BioSum translates into FVS stand files. FVS then simulates multiple, alternative, user-designed silvicultural sequences of up to 4 treatments, implemented at 10-year intervals, interleaved with stand projection between treatments. BioSum then imports FVS output, and sends it to both OpCost for simulating treatment costs for each decade for each

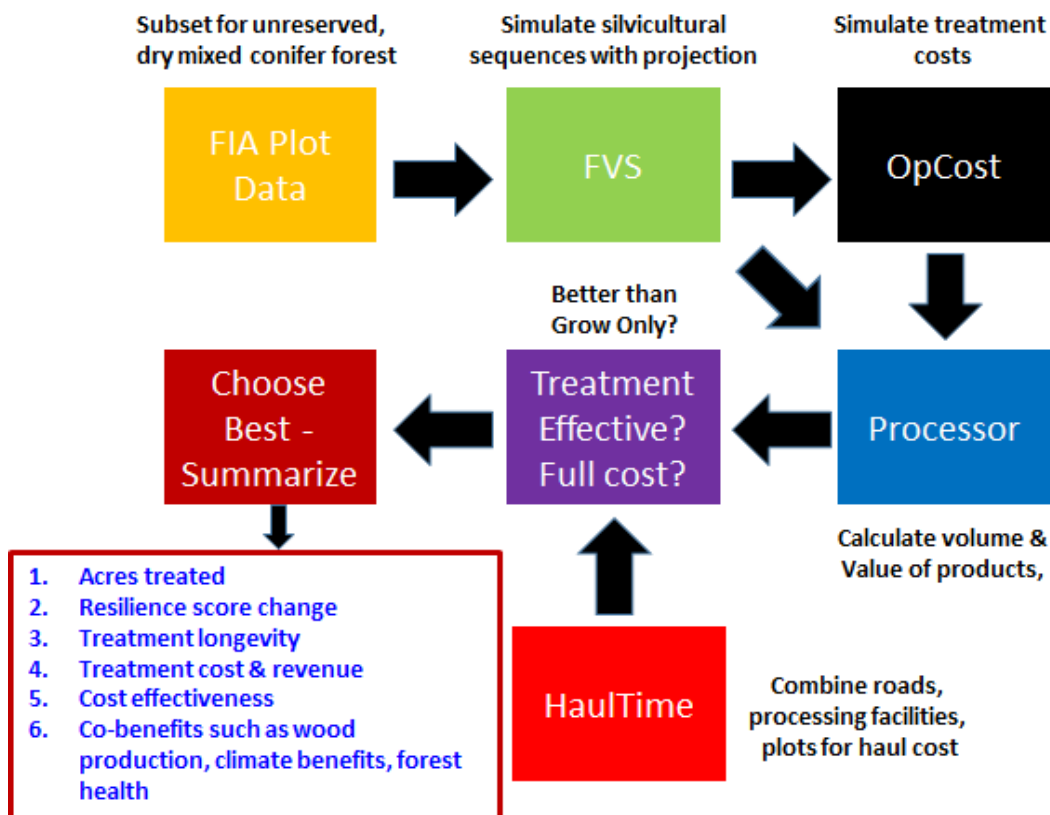


Figure 1—Data and processing workflow within the BioSum analysis framework.

stand-sequence combination and to “Processor” which accounts for wood product volumes and values.

OpCost manages over 100 equations covering 11 types of logging machinery and 11 harvest systems composed of multiple machines (Bell and others, 2017a). Predictions from all applicable equations for a stand, given the selected harvest system, are generated and averaged to obtain a treatment cost. Validation of OpCost predictions has been published (Bell and others 2017b) and work on implementing a harvest system optimization option in BioSum and OpCost is continuing.

Next to enter the workflow are travel times from every plot to every potentially relevant processing facility estimated via R (R Core Team 2017) scripts that implement a graph theory representation of the road network. Ultimately, we must define what an effective management sequence looks like, and how to choose the best one when there are several candidates. This numbered list shows a few of the kinds of summaries that can emerge from the end of the pipe. Because we project stand trajectories following treatment, we can address treatment longevity directly.

USING BIOSUM

There is no one correct way to use BioSum. We and research partners have used BioSum to, for example:

1. Assess the status of and opportunities to achieve risk reduction and other goals in current forests
2. Apply silvicultural prescriptions today, and monitor how effects play out over time
3. Simulate dynamic management over four projection cycles
4. Evaluate outcomes of silvicultural alternatives over a wide range of possible options, in order to rate or rank them by appropriate metrics
5. Predict and evaluate the product mix that forested landscapes can produce under different policies, legal and economic restrictions, or incentives
6. Convert FIA data into FVS format to assess or experiment with stand data from a representative sample of the forested landscape

The illustrative example presented here can be thought of as a blend of uses: assessment (#1), silvicultural prescription scenarios (#2) and effectiveness (#4). Through this proactive monitoring analysis, BioSum provides an initial, model-informed test of a hypothesis designed to evaluate alternative management choices. Over time, the continuous remeasurement of the FIA sample plots offers the opportunity to obtain monitoring feedback about the real world outcomes of such management, assuming that implementation actually happens at a scale sufficient for detection by the FIA plot network. This can be best seen as a supplement to stand-to-landscape effectiveness monitoring that is needed to judge outcomes of particular implementations in particular places to promote learning, inform future management decisions, and improve model accuracy.

FUEL TREATMENT EXAMPLE

To illustrate one use of the framework, we looked at the effectiveness and costs of mechanical fuel treatments designed to reduce fire hazard and enhance fire resistance, focusing on dry mixed conifer forests across the geographic range of 13 FVS variants in CA, OR, WA, ID and MT (FVS version 1778). This FIA sample represents 29 million acres with over 7,000 conditions (full or partial plots). By applying the BioSum analysis framework, these conditions become stands that get modeled in FVS. These stands cover almost every gradient imaginable, across density, volume, site quality, age, structure complexity, species fire tolerance, terrain, road access, and proximity to wood processing facilities. Where a stand sits in this hyperspace determines its inherent resistance, amenability to restoration treatment, longevity of treatment benefits, and net treatment costs or revenues.

Relying on the FVS Structural Statistics Report as a basis for characterizing forest structure and drawing on prescription examples shared during interviews with silviculturists across the region, three stand types were recognized: (1) multi-storied stands, for which we devised six versions of an “improvement cut” prescription designed to maintain multi-storied stand structure while reducing overstory canopy density and understory tree count; (2) single story stands, which we addressed with three versions of

a “commercial thin” prescription, and (3) young stands containing trees too small to be suitable for either of these kinds of prescriptions, which we did not model for this study. Table 1 shows ranges of key prescription parameters. For both multi- and single-storied stands, prescriptions were designed to first cut low vigor trees (those with live crown ratio < 40 percent or height to DBH ratios exceeding 80), then cut tree species considered not resistant to fire, such as white and grand fir, then additional trees until prescription targets were achieved, subject to specified DBH ranges. Mechanized whole-tree logging was modeled on slopes under 40 percent and cable manual whole-tree logging on steeper slopes to minimize generation of in-forest residues; such residues were piled and burned only when they resulted in surface fuels exceeding 15 tons/acre as simulated in FVS. Post-treatment regeneration was added using the REPUTE model. Grow-only simulations provide a baseline against which to compare the stand trajectories achieved via active management.

Treatment Effectiveness

BioSum analyses have long relied on metrics produced by FFE-FVS, such as torching and crowning indices, torching probability, surface flame length and derivatives of predicted fire-induced mortality volume as indicators of hazard, and on changes in such metrics as a measure of effectiveness. However, experience has demonstrated that FFE-FVS metrics are driven much more by surface fuel model choices than tree attributes, and despite years of effort to finesse FFE-FVS’s fuel model selections, confidence that model outcomes are realistic has been elusive. Instead, we

derive resistance metrics from tree information—the kind of information that FIA plots most reliably provide.

We used four management approaches to increasing stand resistance to fire: (1) elevating canopy base height, (2) reducing canopy bulk density, (3) increasing proportion of resistant species, and (4) increasing tree size (Agee and Skinner 2005). We did not model surface fuel trajectories in this analysis, but accounted for surface fuel treatment cost and implicitly addressed surface fuels by developing a target canopy base height (CBH) metric (Keyes 2006, Keyes and O’Hara 2002). Each of these four dimensions of resistance was scored (0-3) to produce a component resistance metric (CRM). These were ultimately summed to calculate a composite resistance score (0-12) to integrate across these factors. Keeping large trees alive, harvesting and sequestering woody carbon in products, and utilizing residues for renewable energy all contribute to GHG mitigation, an important co-benefit.

To consider **target CBH**, all relevant timber litter and timber understory fuel models (Scott and Burgan 2005) were modeled in BEHAVE under a broad range of wind speeds and slopes to derive intensity and generate inputs for the van Wagner equation (van Wagner 1977) that calculates the target canopy base height required to prevent crown fire initiation. While these target CBHs vary with wind and slope, as well as fuel, we observed some clustering and natural breakpoints that suggested suitable thresholds for scoring this CRM: 0 for CBH < 7 feet, 1 for $7 \leq \text{CBH} < 20$, 2 for $20 \leq \text{CBH} < 30$ and 3 for $30 \leq \text{CBH}$.

Table 1—Silvicultural prescription parameters used to define 6 “improvement cuts” applied to multi-storied stands and 3 “commercial thins” applied to single story stands.

Treatment	Residual stand basal area or trees per acre (TPA) target	Max DBH (inches)	Min DBH (inches)	Understory Target TPA
Improvement cuts	80 to 100 ft ²	19-21, none	5-7	0 to 222
Commercial thins	150 ft ²	None	7	50
	90-194 TPA	None	5-7	20

We relied on the literature to score resistance conferred by **canopy bulk density** (CBD) as follows: 0 for $CBD > 0.15 \text{ kg/m}^3$, 1 for $0.1 < CBD \leq 0.15$, 2 for $0.05 < CBD \leq 0.1$, and 3 for $CBD \leq 0.05$. A stand scoring zero for this CRM has essentially no resistance to active crown fire propagation, while one earning a 3 not only has considerable resistance, but can grow for a while before resistance fades.

Western larch, ponderosa pine, Jeffrey pine, sugar pine, and red fir are considered fire resistant species in all 13 variants, and Douglas-fir in all except the Inland Empire, Blue Mountains and Eastern Montana variants. We calculated **resistant species proportion** (prop) as a fraction with numerator containing the basal area of all live trees of species that are considered fire resistant in that variant and denominator containing the basal area of all live trees. Scoring of this CRM was as follows: 0 for $\text{prop.} < 0.25$, 1 for $0.25 \leq \text{prop.} < 0.50$, 2 for $0.50 \leq \text{prop.} < 0.75$, and 3 for $0.75 \leq \text{prop.}$

Accounting for the tree size component of fire resistance, intended as a proxy for survival of live trees, was complicated by the simultaneous effects of size and species on survival. Mean DBH, height and crown ratio for all the trees in the FIA database were calculated, by species, size class and FVS variant to produce inputs for the First Order Fire Effects Model (FOFEM), version 6, which was used to predict mortality resulting from 6 and 8 foot flame lengths for each species-size class-variant combination. The species-size class-variant appropriate mean (of 6 and 8 foot flame length predictions) for each combination was applied to the trees per acre (TPA) represented by each live tree as a mortality factor, and these were used to expand tree volume ($\text{mortality TPA} * \text{volume}$) to mortality volume. Mortality volume was summed over all trees, then used to compute **survival proportion** as $((\text{TPA} * \text{Volume}) - \text{mortality volume}) / (\text{TPA} * \text{Volume})$. Proportions were scored as follows: 0 for < 0.02 , 1 for $0.02 \leq \text{prop.} < 0.30$, 2 for $0.30 \leq \text{prop.} < 0.60$, and 3 for $0.60 \leq \text{prop.}$ This scoring awards a point for even very minimal proportional survival. When a stand contains trees that are of a size and species that result in 60 percent volumetric survival, this system considers the stand fully resistant with respect to this CRM.

These four CRMs were summed to produce a **composite resistance score** (CRS) that ranges from 0 to 12. This score can be calculated for pre- and post-treatment time points or for any other time point in the simulation. We can compare CRS at a particular time, or as a weighted average over a period of time, that results from one treatment versus another or to a grow-only scenario. In this way, treatment longevity can be explicitly considered in the analysis framework, and the effects of intentional management separated from changes that might occur anyway with natural succession in the absence of management.

Classifying Fire Vulnerability

Exploratory analysis of these calculated metrics (CRS and CRM) for thousands of stands revealed some distinctly different initial (pre-treatment) conditions that we believe are germane to identifying superior management alternatives. We constructed four bins, which we'll refer to as fire vulnerability classes (FVC), to partition the range of resistant species proportion, as this metric appears to strongly influence the potential for treatments to be effective (table 2). For example, a stand of pure white fir (FVC 4) cannot be immediately converted to a CRS score of 12 because its low resistant species proportion can't be changed without totally replanting the site. The FVCs also differ in terms of their resistance (as measured by mean CRS) and their prevalence in dry mixed conifer forests. Moreover, their potential for resistance improvement with management differs markedly, as seen for target CBH (fig. 2.). In stands with the lowest fire vulnerability (FVC 1), where CRS is high before any treatment, we see minimal improvement to that component resistant metric from applying restoration treatments. However, treating stands that have a high proportion of resistant species but lower scores for the other metrics (FVC 2) leads to outcomes of elevated target CBH scores that predict enhanced resistance relative to stands classified as FVC 3 or 4, perhaps because the latter contain shade tolerant species more likely to adversely influence this metric as regeneration commences.

Because every stand in a BioSum analysis is tied to a representative location on the ground, and the forest type, owner, and myriad site factors

Table 2—Pre-treatment fire resistance can be usefully classified or binned into fire vulnerability classes (FVCs) that partition the range of resistant species proportion

FVC	FVC description	Resistant species score	CRS Limit	Percent of forest	Mean CRS
1	High resistance sp. + high total score	3, $\geq 75\%$ fire resistant spp.	≥ 9	19	10.1
2	High resistant sp. + low total score	3, $\geq 75\%$ fire resistant spp.	< 9	10	7.3
3	Mod. resistant sp.	1 or 2, 25-75% fire resistant spp.	All values	33	7.4
4	Low resistant sp.	0, $< 25\%$ fire resistant spp.	All values	37	5.1

CRS=Composite resistance score.

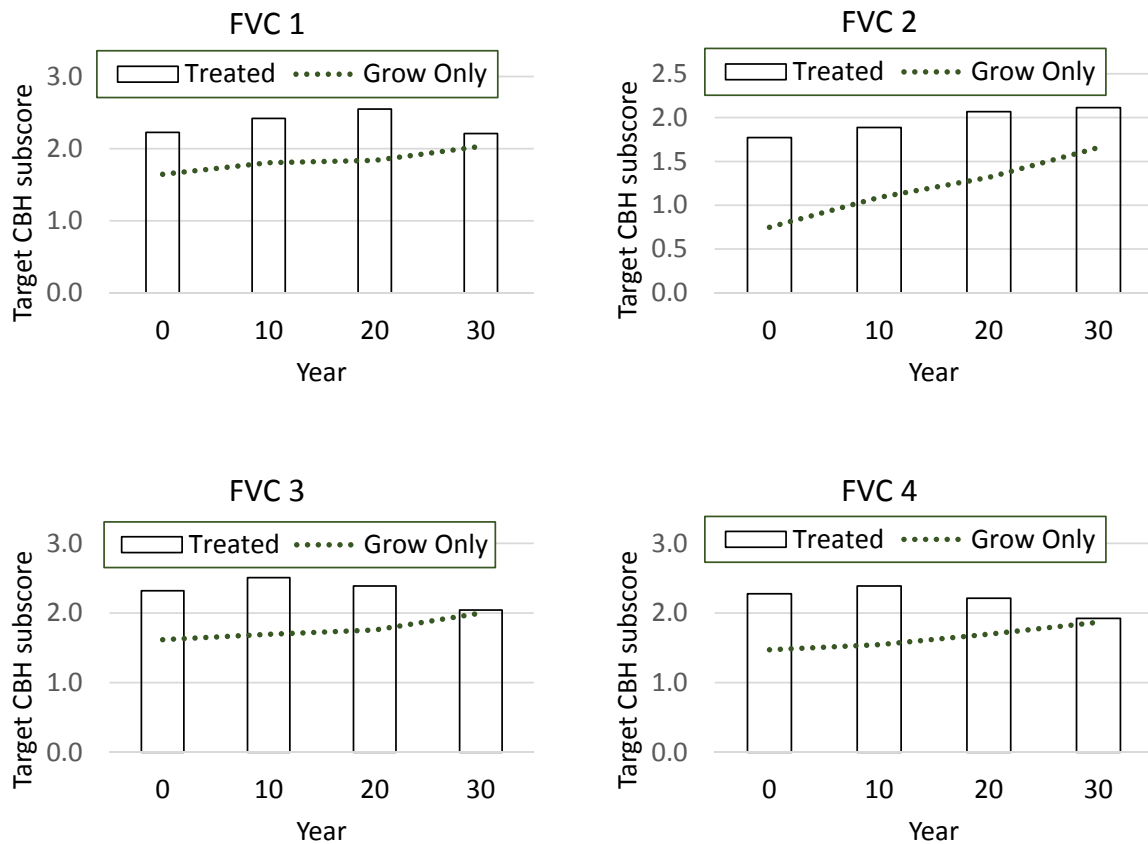


Figure 2—Mean target canopy base height subscore, where 0= less than 7 feet, 1=7 to 20 ft., 2=20-30 ft. and 3= greater than 30 feet., by fire vulnerability class (FVC), where FVC 1=high resistant species sub-score and composite resistance score, FVC 2= high resistant species sub-score and low to moderate composite resistance score, FVC 3= moderate resistant species sub-score, and FVC 4= low resistant species sub-score, when most effective treatment was applied (hollow bars) and when no treatment was applied (dotted lines) over three decades.

associated with that location, it's easy to use these factors as a basis for summarizing any stand level metric collected by FIA or computed in FVS or FFE-FVS, or in this case, via FVC assignment derived from a complex resistance rating process that builds on attributes from those models as well as exogenously calculated information (on survival proportion). Figure 3 shows pre-treatment FVC distribution for dry mixed conifer forests to be highly varied across the National forests in the western portion of the study area, with Lassen having the lowest, and Siskiyou and Six Rivers the highest proportion of area with the highest level of resistant species proportion (FVCs 1 and 2). Reasons for these differences can be hypothesized and tested via analysis of the underlying inventory data.

Treatment Longevity

Comparing the average outcomes of implementing for each stand the restoration treatment that achieves the greatest increase in CRS over the grow-only at each time step confirms that the already high CRS-scoring stands in FVC 1 show less improvement over time when compared to the

grow-only (fig. 4). Three decades after treatment, the gains in average resistance conferred by restoration relative to grow-only scenarios for stands in FVC 1 have completely disappeared. Additional work is underway in a related study to examine re-treatment efficacy and feasibility.

Treatment Economics, Effectiveness and Feasibility

A key BioSum strength is support for scenario analysis, considering, for example, alternative policies and constraints that govern which acres would be prioritized over the forested landscape, given the outcomes of restorations treatments and their net cost, as assessed via net revenue (NR). Four simple scenarios involving differences in the magnitude of the difference in scores (ScoreDiff) between the best restoration treatment and grow-only sequences and levels of treatment subsidy that can be contemplated, and considering only the ScoreDiff at year 1, were evaluated to produce the comparison of outcomes depicted in figure 5 with respect to area treated, mean net revenue and mean ScoreDiff. The scenarios are:

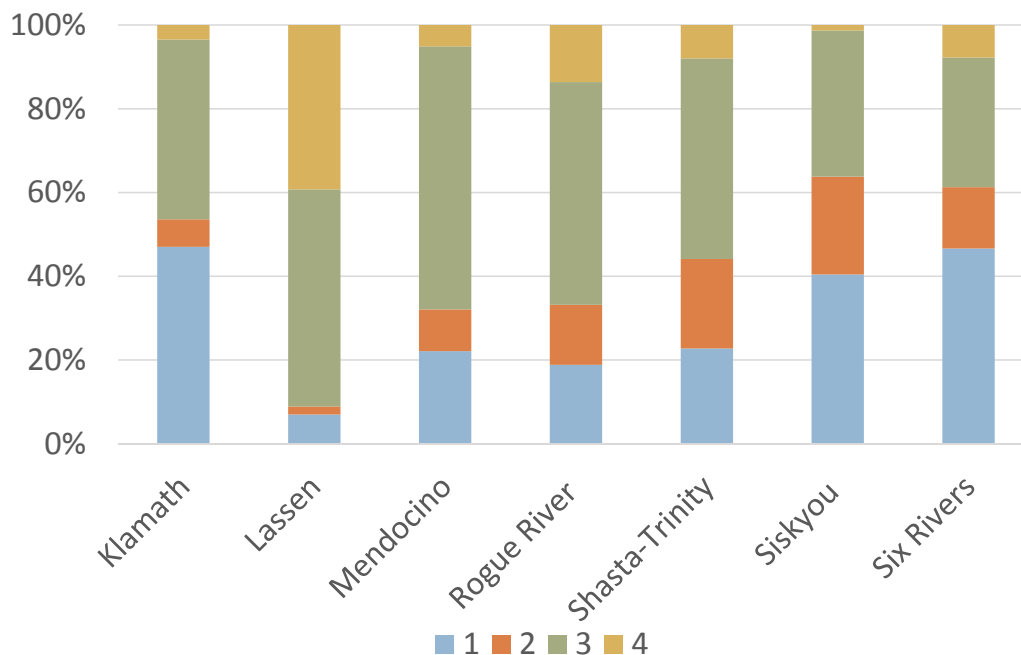


Figure 3—Distribution of forest area, as a percent of total area, by fire vulnerability class (FVC) for seven national forests in the western portion of the study area, where FVC 1=high resistant species sub-score and composite resistance score, FVC 2= high resistant species sub-score and low to moderate composite resistance score, FVC 3= moderate resistant species sub-score, and FVC 4= low resistant species sub-score.

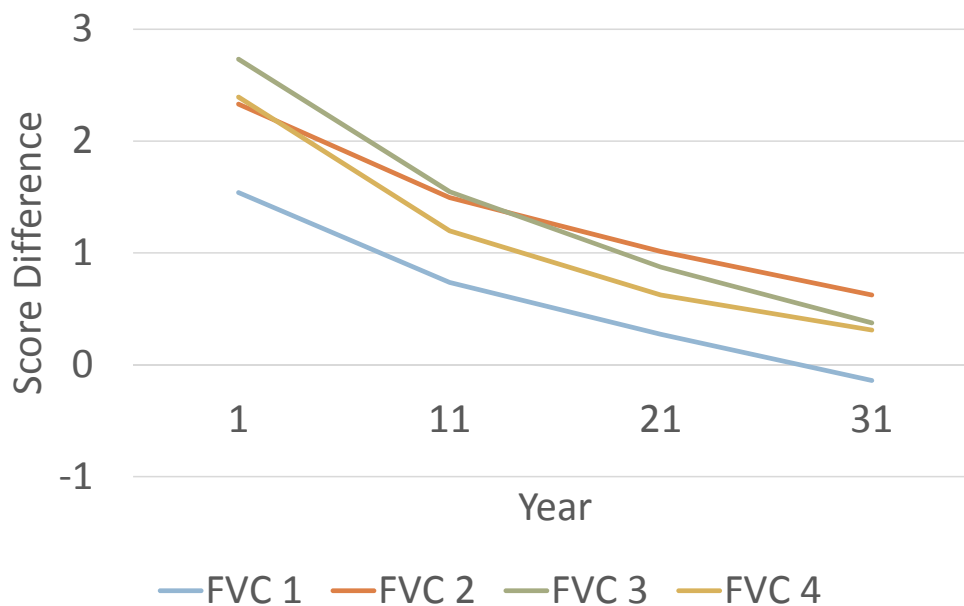


Figure 4—Mean fire resistance score difference in the 12-point scale composite resistance score, relative to a grow-only scenario, by fire vulnerability class (FVC) and decade, where FVC 1=high resistant species sub-score and composite resistance score, FVC 2= high resistant species sub-score and low to moderate composite resistance score, FVC 3= moderate resistant species sub-score, and FVC 4= low resistant species sub-score.

1. Score improves by at least 1 point (ScoreDiff>0, since scores are integers)
2. Score improves by at least 1 plus treatment pays for itself (ScoreDiff>0, NR >0)
3. Score improves by at least 1 and net treatment costs are between 0 and \$500 per acre (ScoreDiff>0, NR 0 to -500)
4. Score improves by at least 3 and net treatment costs are between 0 and \$500 per acre (ScoreDiff>2, NR 0 to -500)

Restoration treatment has the potential to at least somewhat increase resistance, at least initially, on approximately 17 million acres of dry mixed-conifer forest in this five State region; however, self-paying treatment is possible on only about half of that area (fig. 5A). As seen earlier, resistance improvement, as measured by ScoreDiff, in FVC 1 stands is somewhat less than for stands in the other classes (fig. 5B), and the mean improvement is somewhat less for stands where subsidy is required (NR of 0 to -500). However, for about a third of these stands, a ScoreDiff of 3 or greater can be attained, and at a unit cost about equal to the average for

the full set of NR 0 to -500 stands, which suggests opportunities to prioritize—using the first available funds to treat acres with greater ScoreDiff. Most of the acres with negative net revenue would require subsidies greater than \$500 per acre (compare a sum of the 2nd and 3rd bars with the 4th in fig. 5A) to achieve a significant reduction in fire vulnerability.

Although most restoration treatments incur net costs, even after accounting for sales of wood produced, the revenue from those that produce positive net revenue is large enough that addressing all treatable acres would generate positive cash flow, except for stands in FVC 4. Unsurprisingly, limiting treatment to stands that pay for themselves generates much more revenue per acre, but treats much less area, though the improvement on acres that are treated is not dramatically different with or without such limits (fig. 5B, 5C). A caveat on the economic analysis is that only treatment and haul costs are considered; administrative and planning costs are not included in the estimates. It is hoped that implementation of BioSum would increase the transparency and accuracy of planning, with the potential to reduce planning costs.

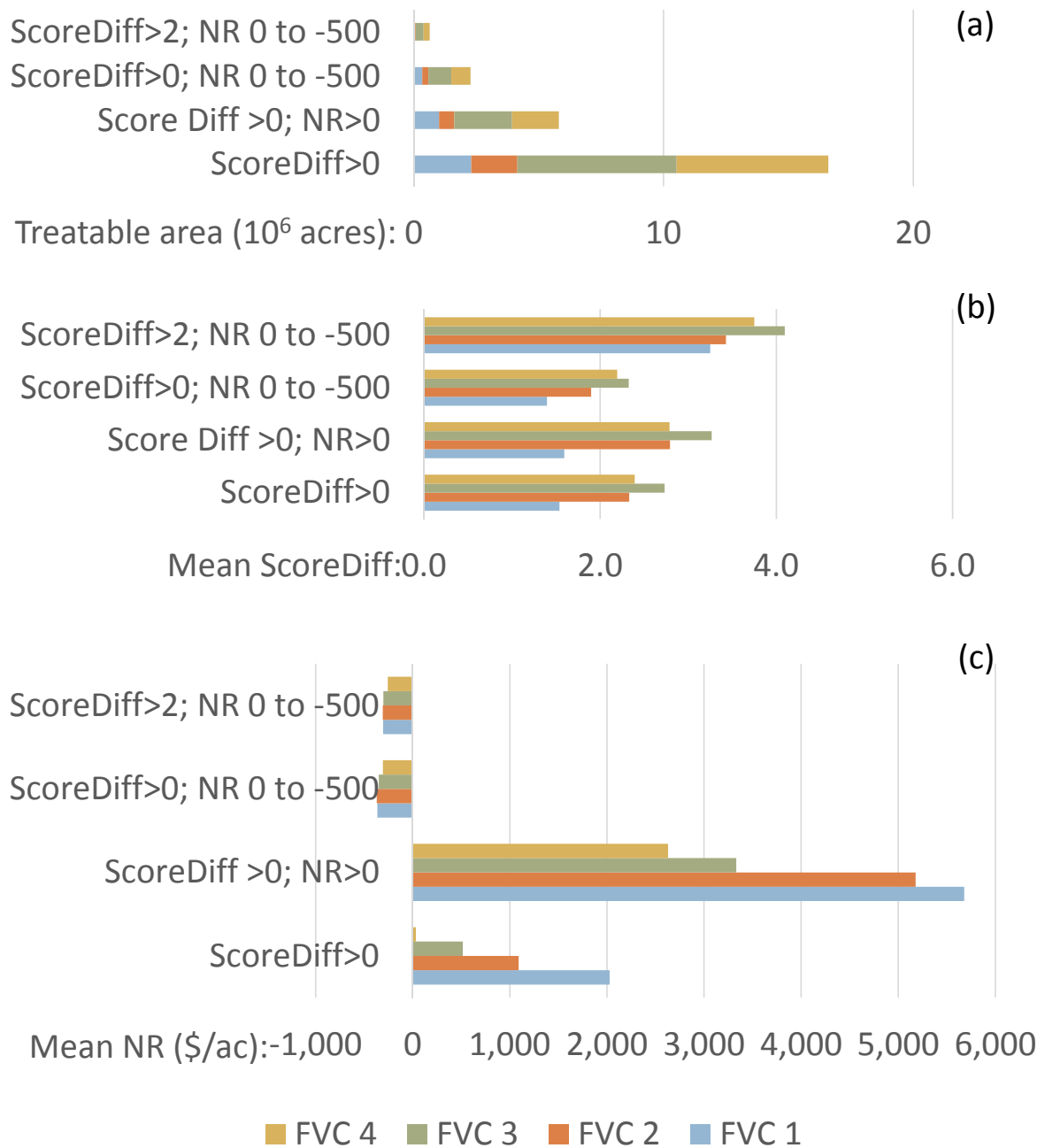


Figure 5—Area treated, in millions of acres (a); mean difference in composite resistance score (ScoreDiff) at year 1 between applying the most effective treatment and no treatment (b); and mean net revenue, in dollars per acre, of applying the treatment that generates the greatest increase in resistance score (c), by pre-treatment fire vulnerability class (FVC) under four scenarios: 1. Score improves by at least 1 point (ScoreDiff > 0), 2. Score improves by at least 1 and treatment pays for itself (ScoreDiff > 0, NR > 0), 3. Score improves by at least 1 and net treatment costs are between 0 and \$500 per acre (ScoreDiff > 0, NR 0 to -500), and 4. Score improves by at least 3 and net treatment costs are between 0 and \$500 per acre (ScoreDiff > 2, NR 0 to -500).

MONITORING PROSPECTS

BioSum and the FIA plot network have potential utility for monitoring the outcomes of forest restoration implementation. BioSum analyses like this one can provide at least preliminary, model-based information about the likely outcomes of alternative management choices and about prospects for long-term success. However it is important to remember that, provided that the program remains funded, the FIA data will continue to roll in, so if implementation of those management choices produces substantial changes on the landscape, this becomes visible as the data updates and it will be possible to validate whether the forested landscape is changing as desired. If managed area is not large, there may be value for National forests in analyzing an overlay of treatment polygons in enterprise databases such as FACTS on FIA plot locations, provided that treatment polygons can be consistently populated and updated—something we have not yet found to be universally true.

AVAILABLE NOW

A forthcoming article (Fried and others 2017) more fully describes the BioSum framework and other examples of analyses conducted to date. This, and other BioSum related publications and the BioSum software and Users Guide, can be downloaded from <http://biosum.info> at no charge. FIA program data to feed BioSum can be downloaded from https://apps.fs.usda.gov/fia/datamart/datamart_access.html.

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Computational Techniques

An Evaluation of CLIMATE Site Index in Large-Tree Diameter Growth Modeling of Selected Tree Species in the Great Lakes Region, U.S.A.

Ram K. Deo, Robert E. Froese, Matthew B. Russell, and Michael J. Falkowski¹

Tree growth models are instrumental in stand growth and yield projection frameworks such as the Forest Vegetation Simulator (FVS) that widely integrates site index (SI) to account for the influence of forest productivity on stand dynamics (Dixon 2002). The models for species-specific SI are traditionally obtained from total height and age measurements of sample trees selected from competition free sites in even-aged stands (Pokharel and Froese 2009). The estimates of SI are prone to error due to inaccuracy in tree age and height data, particularly for diffuse porous shade tolerant species in mixed species natural stands (Froese and Robinson 2007). Because site productivity depends on the interaction of numerous biotic and abiotic factors, estimates of SI obtained at limited sampling locations can be coupled with freely available multiple geo-climatic spatial grid layers to predict SI over a large-area for a wall-to-wall coverage (Monserud and others 2008, Weiskittel and others 2011). The spatially predicted SI can avoid operation difficulty associated with deriving SI empirically, and potentially can substitute the estimated SI as an input to tree growth models. This study integrated SI estimates of the U.S. national Forest Inventory and Analysis (FIA) plots with a suite of co-located spatial grid metrics consisting of temperature, precipitation, soil, and canopy reflectance properties in a non-parametric random forest modeling framework (Deo and others 2016) to produce SI maps for five major species (i.e., red pine, northern white cedar, sugar maple, quaking aspen, and northern red oak) in the Great Lakes region consisting of the States of Michigan, Minnesota, and Wisconsin. The main objective was

to evaluate alternative ways of including site factors in the formulation and application of species-specific large-tree diameter growth models. The performance of spatially predicted SI was tested in newly formulated large-tree (≥ 12.7 cm diameter at breast height DBH) diameter growth models for the same five species. The predictors used in the growth models included initial tree size and competition variables, and the three alternatives for the site factor (Deo 2014) as in the equations 1, 2, and 3, respectively. As an attempt to decouple growth models from the error-prone SI estimate, an approach of directly including geo-climatic variables in the models was tested with the ultimate goal of improving accuracy and making the models sensitive to climate. We have formulated three types of growth equations for each of the species and evaluated their performance in terms of growth prediction accuracy.

$$\ln DDS = \beta_1 + \beta_2 \cdot \frac{1}{D} + \beta_3 \cdot D + \beta_4 \cdot D^2 + \beta_5 \cdot \frac{D}{QMD} + \beta_6 \cdot \frac{D^2}{QMD} + \beta_7 \cdot SBA + \beta_8 \cdot BAL + \beta_9 \cdot CR + \beta_{10} \cdot CR^2 + \beta_{11} \cdot SI_{fia} \quad (1)$$

$$\ln DDS = \beta_1 + \beta_2 \cdot \frac{1}{D} + \beta_3 \cdot D + \beta_4 \cdot D^2 + \beta_5 \cdot \frac{D}{QMD} + \beta_6 \cdot \frac{D^2}{QMD} + \beta_7 \cdot SBA + \beta_8 \cdot BAL + \beta_9 \cdot CR + \beta_{10} \cdot CR^2 + \beta_{11} \cdot SI_{spatial} \quad (2)$$

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$$\ln DDS = \beta_1 + \beta_2 \cdot \frac{1}{D} + \beta_3 \cdot D + \beta_4 \cdot D^2 + \beta_5 \cdot \frac{D}{QMD} + \beta_6 \cdot \frac{D^2}{QMD} + \beta_7 \cdot SBA + \beta_8 \cdot BAL + \beta_9 \cdot CR + \beta_{10} \cdot CR^2 + \beta_{11} \cdot DD + \beta_{12} \cdot MAP \cdot DI + \beta_{13} \cdot MAP + \beta_{14} \cdot DI + \beta_{15} \cdot GSP + \beta_{16} \cdot PI + \beta_{17} \cdot MNDVI + \beta_{18} \cdot MTWM + \beta_{19} \cdot MAT + \beta_{20} \cdot BAWHT \quad (3)$$

where

DDS= 10 years difference in over-bark diameter squared (cm²)

DBH = diameter at breast height (cm)

QMD= quadratic mean diameter (cm)

CR = crown ratio

SI = site index (m)

BAL= basal area of larger tree than the subject tree (m²ha⁻¹)

SBA = stand basal area (m²ha⁻¹)

MAP = mean annual precipitation (mm)

DI = soil drainage index

PI= soil productivity index

GSP = growing season precipitation (mm)

DD = degree-days above 5⁰C accumulating within the frost-free period

MNDVI= MODIS sensor derived normalized difference vegetation index

MTWM= mean temperature in warmest month (°C)

MAT= mean annual temperature (°C)

BAWHT= basal area weighted canopy height (m)

MAP·DI = interaction of mean annual precipitation and soil drainage index

β_i = species dependent regression coefficients.

The reference frame for model training were created using live tree data from two consecutive cycles of FIA measurements so that each tree had an identifier, diameter increment in 10-year period,

initial DBH, and other derived size and competition related variables. The climate, soil and remote sensing variables were also attached to each tree using the fuzzed and swapped coordinates of the FIA plots (Woudenberg and others 2010). The reference frame was divided into two halves; the first half was used to develop the species-specific SI models so that the second half had spatially predicted SI (SI_{spatial}), FIA estimated SI (SI_{fia}) and the geo-climatic variables attached to each tree. The second half was used to develop the three forms of diameter growth equations, following the stepwise and best-subset method of multiple linear regressions (Deo and others 2016, 2017). The growth equations were applied to an independent dataset from the Bureau of India Affairs (BIA) that has established reservation plots for continuous forest inventory.

The direct combination of geo-climatic variables in the growth models improved fit statistics compared to the models using SI_{fia} or SI_{spatial} (table 1). The success of SI_{spatial} was either similar to or worse than the SI_{fia} and varied with species. The sensitivity analysis and importance ranking of the predictors revealed similar ranks of SI_{fia} and SI_{spatial} in the growth models of red pine and sugar maple (table 2). The models based on SI_{fia} and SI_{spatial} explained similar amount of variance for red pine and sugar maple and the largest drop in adjusted R² was observed for red oak (table 1). The coefficients of SI in the growth models were positive for all the species, except northern white cedar. The negative coefficient of SI_{spatial} for northern white cedar implies that the spatial SI model is unreliable; however, this can be attributed to the characteristic that the species grows over a wide range of sites, remain suppressed for several years and respond quickly to release operation at any age (Bouffroy and others 2012). The best fit models were obtained with red pine while quaking aspen models had the poorest fit. However, it is likely that the model fit statistics can be improved if actual coordinates (against the fuzzed and swapped) of the FIA plots are used to attach the spatial predictors to the size attributes of target trees because soil properties can significantly change with the swapping and fuzzing of tree locations.

Table 1—Coefficients and fit statistics of species-specific diameter growth models in the three forms (equations-1, 2, and 3) with lnDDS as the response variable and measured SI (SI_{ra}), imputed SI ($SI_{spatial}$), and biogeoclimatic (BGC) predictors successively substituting the site variable

Parameters	Red pine models			N. white-cedar models			Sugar maple models			Quaking aspen models			N. red oak models		
	Eqn1	Eqn2	Eqn3	Eqn1	Eqn2	Eqn3	Eqn1	Eqn2	Eqn3	Eqn1	Eqn2	Eqn3	Eqn1	Eqn2	Eqn3
Intercept	5.0194	5.2453	2.8638	1.2901	1.4230	2.9712	4.0513	3.9729	2.6350	5.0433	5.2008	0.4251	2.4519	3.0306	5.6012
1/DBH	-27.7165	-27.1814	-27.7361	0.1056	0.1145	0.1040	-20.4084	-20.2670	-20.1392	-18.8786	-18.2105	-17.9302	0.0805	0.0798	0.0677
DBH							0.0787	0.0826	0.0780				-0.0003	-0.0003	-0.0002
DBH ²							-0.0006	-0.0006	-0.0006				0.4500	0.3358	0.6393
DBH/QMD	0.5659	0.6067	0.6695				-0.5612	-0.5985	-0.4886				-0.0111	-0.0075	-0.0117
DBH ² /QMD	-0.0097	-0.0105	-0.0133	-0.0251	-0.0290	-0.0247	-0.0278	-0.0266	-0.0246	-0.0092	-0.0079	-0.0044	-0.0214	-0.0177	-0.0121
SBA	-0.0151	-0.0134	-0.0091	-0.0038	-0.0040	-0.0030	-0.0020	-0.0024	-0.0026	-0.0046	-0.0053	-0.0063	0.0075	0.0001	-0.0041
BAL	-0.0082	-0.0092	-0.0081				0.0228	0.0228	0.0234	0.0228	0.0246	0.0295			
CR	0.0238	0.0241	0.0279	0.0299	0.0292	0.0281	-0.0001	-0.0001	-0.0001	-0.0001	-0.0001	-0.0002	0.0075	0.0001	0.0067
CR ²	-0.0001	-0.0001	-0.0001	-0.0002	-0.0002	-0.0002	0.0172	0.0179	0.0179	0.0206	0.0088	0.0004	0.0373	0.0155	
SI	0.0189	0.0045		0.0179	-0.0026										
DD5							-0.0004		0.0017			-0.0004			0.0010
MAP/DI							0.0000		0.0000			0.0000			0.0000
DI							-0.0132		0.0179			-0.0065			-0.0168
MAP															-0.0030
MTWM												0.0288			-0.0207
MAT							0.0199		-0.0265			-0.0113			0.0099
GSP							-0.0195		-0.0067			-0.0013			0.0034
BAWHT									0.0006						
MNDVI												0.8550			
PI															0.0121
Adj. R ²	0.47	0.47	0.52	0.41	0.40	0.43	0.40	0.40	0.42	0.25	0.24	0.28	0.45	0.43	0.48
RSS	2845.9	2880.9	2568.9	4104.4	4144.9	3978.1	5727.1	5722.4	5608.3	3170.9	3212.7	3049.1	1218.03	1267.29	1147.13
DF	7914	7914	7910	9898	9898	9893	10565	10565	10561	9262	9262	9256	3096	3096	3088
F-stat	891.2	868.3	728.9	1142.1	1114.8	670.9	798.6	800.2	581.6	528.4	501.4	305.4	363.7	332.4	192.5
AIC	14392.29	14489.31	13588.92	19399.09	19496.40	19099.42	23547.14	23538.40	23333.47	16378.05	16499.34	16026.54	5923.11	6046.17	5752.95

RSS: residual sum of square; DF: degrees of freedom; AIC: Akaike Information Criterion. DBH=diameter at breast height; SBA = stand basal area; BAL= basal area of larger tree; CR = crown ratio; SI = site index; DI = soil drainage index; MAP = mean annual precipitation; MTWM= mean temperature in warmest month (°C); MAT= mean annual temperature (°C); GSP = growing season precipitation (mm); BAWHT = basal area weighted canopy height (m); MNDVI= MODIS sensor derived normalized difference vegetation index.

Table 2—Factor prioritization based on the reduction of variance in the sensitivity analysis of the three forms of growth models

Species	Models	Ranking of predictor variables (in descending order from left to right) by sensitivity analysis												
		CR	1/DBH	DBH ² /QMD	SBA	DBH ² /QMD	BAL	CR ²	S _{fia}	CR ²	DBH ² /QMD	BAL	DD5	
Red pine	Eqn1	CR (0.3891)	1/DBH (0.3863)	DBH ² /QMD (0.0660)	SBA (0.0575)	DBH ² /QMD (0.0503)	BAL (0.0238)	CR ² (0.0138)	S _{fia} (0.0131)	CR ² (0.0204)	DBH ² /QMD (0.0428)	BAL (0.0109)	SBA (0.0098)	DD5 (0.0084)
	Eqn2	CR (0.3982)	1/DBH (0.3703)	DBH ² /QMD (0.0756)	DBH ² /QMD (0.0595)	SBA (0.0453)	BAL (0.0301)	CR ² (0.0204)	S _{fia} (0.0006)	DBH ² /QMD (0.0445)	MTWM (0.0634)	CR ² (0.0307)	SBA (0.0098)	DD5 (0.0084)
	Eqn3	CR (0.2485)	1/DBH (0.1791)	MAPDI (0.1525)	DI (0.1239)	MAT (0.0854)	MTWM (0.0634)	DBH ² /QMD (0.0445)	CR ² (0.0307)	DBH ² /QMD (0.0428)	BAL (0.0109)	SBA (0.0098)	DD5 (0.0084)	
N. white cedar	Eqn1	DBH (0.5997)	CR (0.2327)	DBH ² /QMD (0.1015)	CR ² (0.0591)	S _{fia} (0.0051)	SBA (0.0019)	CR ² (0.0138)	S _{fia} (0.0131)	DBH ² /QMD (0.0445)	MTWM (0.0634)	CR ² (0.0307)	SBA (0.0098)	DD5 (0.0084)
	Eqn2	DBH (0.6312)	CR (0.1989)	DBH ² /QMD (0.1213)	CR ² (0.0466)	SBA (0.0019)	S _{fia} (0.0001)	CR ² (0.0138)	S _{fia} (0.0131)	DBH ² /QMD (0.0445)	MTWM (0.0634)	CR ² (0.0307)	SBA (0.0098)	DD5 (0.0084)
	Eqn3	DBH (0.4971)	CR (0.1759)	DD5 (0.0999)	DBH ² /QMD (0.0837)	MTWM (0.0718)	CR ² (0.0443)	CR ² (0.0159)	MAPDI (0.0024)	BAWHT (0.0027)	MAPDI (0.0024)	SBA (0.0010)		
Sugar maple	Eqn1	DBH (0.5860)	1/DBH (0.1197)	CR (0.0995)	DBH ² (0.0769)	SBA (0.0599)	DBH ² /QMD (0.0383)	CR ² (0.0143)	S _{fia} (0.0045)	BAL (0.0009)				
	Eqn2	DBH (0.6030)	1/DBH (0.1105)	CR (0.0933)	DBH ² (0.0822)	SBA (0.0511)	DBH ² /QMD (0.0407)	CR ² (0.0133)	S _{fia} (0.0047)	BAL (0.0011)				
	Eqn3	DBH (0.4097)	MAPDI (0.1507)	DI (0.1299)	1/DBH (0.0830)	CR (0.0750)	DBH ² (0.0585)	SBA (0.0334)	CR ² (0.0164)	MAT (0.0111)	MAT (0.0111)	GSP (0.0092)	BAWHT (0.0014)	
Quaking aspen	Eqn1	1/DBH (0.4838)	CR (0.3953)	S _{fia} (0.0394)	SBA (0.0368)	CR ² (0.0338)	BAL (0.0108)	CR ² (0.0138)	S _{fia} (0.0045)	BAL (0.0009)				
	Eqn2	CR (0.4546)	1/DBH (0.4478)	CR ² (0.0505)	SBA (0.0267)	BAL (0.0141)	S _{fia} (0.0063)	CR ² (0.0138)	S _{fia} (0.0047)	BAL (0.0011)				
	Eqn3	CR (0.2688)	MTWM (0.2668)	1/DBH (0.1771)	MAPDI (0.0937)	DI (0.0590)	MAT (0.0560)	CR ² (0.0366)	DD5 (0.0172)	GSP (0.0083)	BAL (0.0081)	MNDVI (0.0050)	SBA (0.0034)	
N. red oak	Eqn1	DBH (0.7979)	DBH ² /QMD (0.0667)	DBH ² (0.0382)	SBA (0.0327)	DBH ² /QMD (0.0304)	S _{fia} (0.0248)	CR (0.0093)	S _{fia} (0.0045)	BAL (0.0009)				
	Eqn2	DBH (0.8562)	DBH ² (0.0573)	DBH ² /QMD (0.0334)	SBA (0.0245)	DBH ² /QMD (0.0185)	CR ² (0.0068)	S _{fia} (0.0033)	S _{fia} (0.0047)	BAL (0.0011)				
	Eqn3	DBH (0.4145)	MAPDI (0.1474)	DI (0.1287)	DD5 (0.0600)	DBH ² /QMD (0.0551)	MTWM (0.0536)	D/QMD (0.0452)	MAP (0.0251)	GSP (0.0228)	DBH ² (0.0161)	MAT (0.0145)	SBA (0.0077)	

Note: The numbers in the parentheses represent sensitivity index (i.e., importance rank of the predictors in the model).

We conclude that the accuracy of large-tree diameter growth models can be improved by using geo-climatic variables in the place of FIA estimated SI. We recommend refinements of the models using actual coordinates of the FIA plots, and also a version of FVS that does not require measured SI.

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Adjusting Canopy Cover Estimates for Non-Random Spatial Distributions in FVS

Michael Shettles and Erin Smith-Mateja¹

Estimates of percent canopy cover (PCC) are increasingly used as target metrics in silvicultural prescriptions. The Forest Vegetation Simulator (FVS) projects estimated differences in forest stand development given a range of alternative prescriptions, with PCC being one key metric used to distinguish differences. FVS calculates PCC by using estimated crown width values for individual trees as input into the equation for the area of a circle, resulting in an estimated projected crown area for each tree, assuming symmetric, continuous and circular cover. These individual crown area values are then expanded to a per acre basis, with these expanded values being subsequently summed and related to a single average acre, to arrive at a stand-level value for PCC, uncorrected for crown overlap. Correcting this value for overlap is done using equation 1 from Crookston and Stage (1999), with a key assumption being that trees are randomly spaced. This assumption is reflected in the 0.01 coefficient. This overlap correction factor (OCF) represents random distributions when 0.01, and allows for uniform and clumpy distributions to be represented through an increase or decrease in value, respectively. This default assumption of a random spatial distribution, however, has been observed to produce biased estimates of PCC when trees are from stands with non-random spatial distributions (e.g., clumped or uniform). To assess the magnitude of this bias, Christopher and Goodburn (2008) took a GIS-based approach to assess how different spatial distributions affected the accuracy of FVS estimates of PCC. Using 19 stem-mapped plots and Ripley's $K(d)$ spatial statistic to identify the degree of non-randomness, results showed FVS underestimated PCC by 11 percent for more uniformly distributed stands, and overestimated PCC by 2 percent for clumpy-

distributed stands. Given the array of other stand metrics that can vary for a given PCC estimate (Ecological Research Institute 2012, Sanchez Meador and others 2011), this bias was deemed worth correcting with empirical relationships available for establishment using Forest Inventory and Analysis (FIA) data containing field-measured estimates of PCC.

(1)

$$PCC = 100 * [1 - e^{(-0.01 * UNPCC)}]$$

Using tree-level measurements and the aforementioned estimated values of PCC on 4,599 FIA plots within nine States, relationships between field-measured PCC and FVS-calculated values of uncorrected PCC (UNPCC) were used to develop non-linear regression models for estimating overlap corrections for non-random spatial distributions.

The nine States chosen were selected so as to represent a range of forest types and conditions throughout the Eastern and Western United States. States included in the analysis were: Arizona, Arkansas, California, Kentucky, Montana, Oregon, Pennsylvania, South Carolina, and Wisconsin. State FIA plot data were obtained from available online State databases (FIA Datamart: <https://apps.fs.usda.gov/fia/datamart/datamart.html>) and translated into FVS-ready databases using the FIA2FVS translator program (Vandendriesche 2012). FVS simulations were then conducted for a single cycle, using the appropriate variant covering the geographic areas for all plots. Variants used included: Blue Mountains, Central Rockies, East Cascades, Eastern Montana, Inland California and Southern Cascades, Inland Empire, Klamath Mountains, Lakes States, Northeast, Pacific Northwest Coast, South Central

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Oregon and Northeast California, Southern, Western Sierras and Westside Cascades, with all variants being Version 1615. For use in the forthcoming regression analyses, the FIA field-measured PCC values (FIAPCC) were pulled through to an output database using the SQLIN and SQLOUT keywords which were added to the “Computed Uncorrected Canopy Cover %” keyword component file (i.e., `addfile`)², which computes UNPCC values using an algebraically rearranged equation 1. To prepare the dataset for analysis, both the FIA PCC values and computed UNPCC values were substituted into equation 1, and the OCF values were solved for algebraically. The resultant OCF values ranged from 0.001 to 0.050, and their associated FIAPCC values were then placed into two groups—clumpy and uniform. For OCF values ranging from 0.001 to 0.009, their associated FIAPCC were assumed to be from stands with some degree of aggregation, and those FIAPCC values were thus placed in the clumpy group. The same being done for the FIAPCC values associated with OCF values ranging from 0.011 to 0.5—they assumed were to be from stands with some degree of spatial homogeneity, and were placed in the uniform group. Using the `nls()` package in R statistical software (R Core Team 2016), non-linear least-squares regression was conducted twice to estimate the mean OCF value for the uniform and clumpy groups, whereby the objective function was equation 1, with the OCF being the single unknown parameter to be estimated. From these mean response OCF values, a scale with intermediate degrees of non-uniformity was created by interpolation, resulting in qualitative user-defined ratings with associated OCF values (table 1). These ratings were then integrated into the Suppose graphical user interface for FVS (Dixon 2002).

Mean response OCF values for the clumpy and uniform groups were 0.006035 and 0.015199, respectively. Model root mean square errors for the clumpy and uniform classes were 14.15 and 10.83 PCC, respectively (both p -values $< 2e-16$). The mean OCF values were used for the “Moderately...” qualitative user-defined rating in Suppose (table 1). These results have also subsequently been implemented into the FVS keyword framework. The new keyword, `CCAdj` (corrected percent Canopy

Cover Adjustment), allows users to modify the overlap assumption using these estimated overlap corrections based upon the aforementioned range of user-defined classes on non-uniformity. Values of PCC for non-random distributions can now be used as a target metric in conjunction with the “Thin to a residual percent canopy cover” management action (`ThinCC` keyword). When used to specify post-thin distribution in conjunction with the `ThinCC` keyword (“Thinning to a residual percent canopy cover” management action), the PCC thinning target is calculated using the associated OCF value. This ensures the appropriate amount of crown area is removed during the simulated thinning, resulting in associated changes in all other stand metrics, such as number of trees removed and residual basal area per acre. See table 1 for all ratings and associated OCF values. Additionally, table 1 contains examples of differences in PCC relative to the FVS-default of random spacing, as well as differences in residual basal area and number of trees removed for different OCF values when using the `ThinCC` keyword. This new keyword can also be scheduled conditionally using the FVS Event Monitor (Dixon 2002).

The implementation of the `CCAdj` keyword allows users to modify canopy cover estimates based upon user-specified non-random spatial patterns. While the utility of this is obvious, the onus is, at current, entirely on the user to first define the degree on non-uniformity in their stands for which they are trying to simulate forest management scenarios. It should be noted that changing the OCF value only changes values of PCC, and estimates of growth and mortality remain unchanged. The effects of spatial patterns on these parameters remain separate, and valid, avenues to explore. The next logical step would be to integrate this work with some function, or relatable spatial statistic, to streamline which rating to select, or even possibly refine the resolution of these ratings. Users may want to use the mean response OCF values (e.g., “Moderately...”) if they are sure spacing structure is non-random, but unsure of the degree of such.

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² `addfile` publically available for download at: <https://www.fs.fed.us/fmcs/fvs/software/addfiles.php>.

Table 1—Differences in simulated future estimates of 2024 percent canopy cover (PCC), and both removed trees per acre (TPA) and residual basal area per acre (BA) associated with a scheduled 2014 simulated thinning to a residual 20-percent canopy cover for the range overlap correction factors (OCF)

Spatial distribution	OCF	Removed TPA (2014 ThinCC with 20 PCC Target)	Residual BA (2014 ThinCC with 20 PCC Target)	PCC in 2024
Extremely uniform	0.037703	609	7 ft ² /Ac	30
Very uniform	0.021129	593	12 ft ² /Ac	27
Moderately (Mean) uniform	0.015199	579	17 ft ² /Ac	26
Somewhat uniform	0.011502	562	22 ft ² /Ac	25
Random (FVS Default)	0.010000	552	26 ft²/Ac	25
Somewhat clumpy	0.009296	546	27 ft ² /Ac	24
Moderately (Mean) clumpy	0.006035	501	42 ft ² /Ac	24
Very clumpy	0.003328	395	77 ft ² /Ac	23
Extremely clumpy	0.001301	30	196 ft ² /Ac	22

Thinning simulation was conducted using the Central Rockies variant of FVS, Version 1943. Cutting efficiency=1, Species=All, DBH Range=0-999", Cutting Control=Thin throughout a diameter range.

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Live Tree Carbon Stock Equivalence of Fire and Fuels Extension to the Forest Vegetation Simulator and Forest Inventory and Analysis Approaches

James E. Smith and Coeli M. Hoover¹

The carbon reports in the Fire and Fuels Extension (FFE) to the Forest Vegetation Simulator (FVS) provide two alternate approaches to carbon estimates for live trees (Rebain 2010). These are (1) the FFE biomass algorithms, which are volume-based biomass equations, and (2) the Jenkins allometric equations (Jenkins and others 2003), which are diameter based. Here, we compare FFE and Jenkins-based carbon in aboveground live trees with the component ratio method (CRM) approach (Heath and others 2009) provided in the Forest Inventory and Analysis (FIA) database and focus on identifying where alternate approaches produce equivalent estimates of stand level aboveground live tree carbon.

We have three major objectives in this study where our focus is on the equivalence of alternate approaches when applied to a common set of inventory data:

- (1) Test if estimates of live aboveground carbon stocks produced from the CRM, FFE, and Jenkins methods are statistically equivalent
- (2) Determine if the relative differences between the estimates are consistent across each of the geographic variants, or are variant-specific
- (3) Within variants, identify equivalence or patterns in equivalence by forest type groups and at successively greater levels of aggregations such as all softwood or hardwood forests or whole variants.

We use equivalence testing to address these objectives. Equivalence testing essentially reverses

the burden of proof, based on the idea that failure to reject a null hypothesis does not mean that the null hypothesis is true. So, in contrast to more common approaches to hypothesis testing where the null hypothesis is “no significant difference” the null hypothesis of an equivalence test is “the populations/groups are significantly different.” An overview of equivalence testing can be found in Parkhurst (2001) and Brosi and Biber (2009). An essential feature is that equivalence bounds are set by the investigator to reflect a value that constitutes a meaningful difference. In this case, we test for equivalence defined as a difference between alternate estimates of carbon stock within ± 5 percent or 10 percent of the mean.

Inventory data were obtained from the Forest Inventory and Analysis Data Base (FIADB), which is compiled and maintained by FIA (USDA Forest Service 2016). The specific data in use here were downloaded from <http://apps.fs.fed.us/fiadb-downloads/datamart.html> on May 13, 2016 and include the most recent evaluations—or cycle of the permanent inventory plots across each State—encompassing the conterminous United States plus southern coastal Alaska and measurements obtained on plots from 2004 through 2015. For consistency, only those plots representing a single forested condition are used in FVS simulations (USDA Forest Service 2016). We exclude non-stocked or very young (i.e., under 10 year) plots from the analysis because the lack of trees on these forest plots results in a zero-difference in carbon, an artifact biasing the resampling needed to develop the equivalence tests. We used the

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FIA2FVS utility to produce the required files to run FVS, and conducted FVS runs for each State and variant (variant version number 1778, April 07, 2016) to generate plot-level live aboveground live carbon estimates for all trees ≥ 1 inch diameter at 4.5 foot height using the FFE default and Jenkins methods. Plot level estimates were calculated for CRM (USDA Forest Service 2016) directly from the FIADB.

Equivalence tests presented here are paired-sample tests (Feng and others 2006, Mara and Cribbie 2012), with plot-level pairs on each plot (e.g., CRM and FFE). A distribution of mean difference was obtained through bootstrap resampling. The test statistic is the confidence interval about that distribution of mean difference between paired estimates as applied in two one-sided tests of the null hypothesis (Berger and Hsu 1996). Equivalence—rejection of the null hypothesis that the two approaches are different—is the conclusion when the test statistic (95 percent CI) falls entirely within the specified equivalence threshold (e.g., within ± 10 percent of mean carbon stock). See Hoover and Smith (2017) for expanded presentation of these methods.

We conducted equivalence tests at several levels of aggregation: whole-variant, by hardwood or softwood type groups within each variant, and by the FIA forest type groups within each variant. The Western United States is covered by 15 major FVS variants, each with different parameters and equations, while the Eastern United States is represented by four variants. In some cases, a user's study area may include more than one variant. Examining the mean variant-wide difference between carbon stock estimates calculated by each method (Jenkins minus CRM, Jenkins minus FFE, and CRM minus FFE), there is a general pattern of Jenkins estimates being generally higher than the CRM or FFE estimates, as noted by (Domke 2012), with the CRM and FFE approaches exhibiting the smallest average difference. This is an expected outcome, since both the CRM and FFE methods are based on the volume-to-biomass approach. There is no consistent pattern across variants; while the CRM and FFE estimates are most often equivalent, this is not always true. In some variants, such as Central States, none of the estimates are equivalent,

while all of the estimates are equivalent in the Southern and Klamath Mountains variants, for example.

At the forest type group within variant level, patterns of equivalence are highly variable, with some forest type groups more likely to have at least one pair of equivalent estimates across multiple variants (e.g., lodgepole pine in the West) while other type groups are rarely equivalent (e.g., aspen/birch in the West). In many cases, several different volume equation sets are in use within a variant (fig. 1); part of the variability among forest type groups or variants may be attributed to the many combinations of volume equations underlying the estimates. In general, softwood groups are slightly more likely to have at least one of the pairs of carbon stock estimates identified as equivalent than are the hardwood groups. The paired CRM and FFE approaches more frequently produce equivalent estimates than do the other two paired approaches, but none of these results are consistent across all variants. Each of these results—more common equivalence of softwoods and the CRM-FFE pair—become more apparent at increasing levels of aggregation, particularly in the East (table 1). When comparing carbon stock estimates generated using different methods, scale of the assessment is important to consider because the trend of greater equivalence with aggregation suggests that estimates for larger spatial extents are less sensitive to the choice of estimation method.

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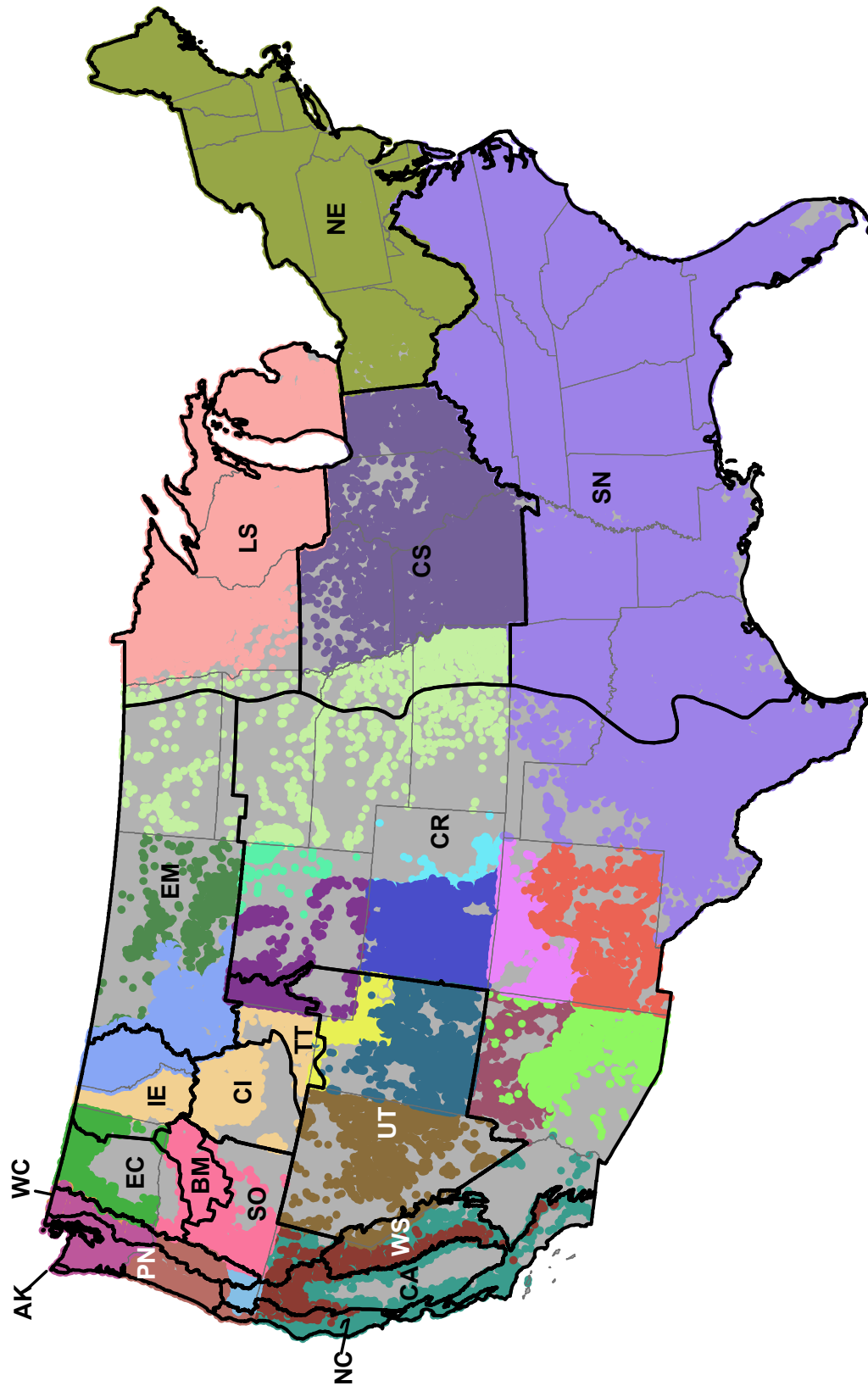


Figure 1—Location of the 19 FVS variants (lines) and the regional volume equations underlying the CRM approach (colors). See FVS or the FIADB documentation for details.

Table 1—Equivalence results from aggregating all western and eastern softwood and hardwood type groups for the three estimation approaches

Estimation approach	Equivalence Level ^a	Western Softwoods ^b	Eastern Softwoods ^c	Western Hardwoods ^b	Eastern Hardwoods ^c
Jenkins- CRM	5%	No	Yes	No	No
	10%	No	Yes	No	No
Jenkins - FFE	5%	No	No	No	No
	10%	No	Yes	No	No
CRM-FFE	5%	No	Yes	No	Yes
	10%	Yes	Yes	Yes	Yes

^a Equivalence levels tested are 5 and 10% of the mean difference between pairs.

^b Western is defined as all other variants, including Alaska.

^c Eastern is defined as the Lake States, Northeast, Central States, and Southern variants.

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Theoretical Foundation of Stage's Formulation of Stand Density Index

Hsien-chih Bryan Lu, Fred Martin, and Ralph Johnson¹

Abstract—Stand density index (SDI) is calculated in all variants of the Forest Vegetation Simulator (FVS) and displayed in output tables. In addition, it is also used to drive the mortality function in a number of variants. Reineke (1933) developed SDI to quantify the relative density of an even-aged stand. Stage (1968) showed that the Reineke's SDI can be computed tree by tree. His SDI formulation was implemented as a Fortran subroutine in FVS. The theoretical foundation of Stage's SDI formulation is revealed in this paper.

INTRODUCTION

Stand density index (SDI) is calculated in all variants of FVS. SDI provides a measure of competition, both between stands and within individual stands, and among groupings of species or sizes within stands. It is also a variable in the mortality function of a number of variants. Hence, it is important to understand how SDI is implemented in FVS.

Reineke (1933) developed SDI to express the density of an even-aged stand. Stage (1968) showed that Reineke's SDI can be alternately computed by summing information from individual trees. However, he did not provide the theoretical foundation for this formulation. Stage's SDI formulation was written as a Fortran subroutine (see <https://sourceforge.net/p/open-fvs/code/HEAD/tree/branches/FMSCrelease/base/src/sdical.f>) in FVS.

Although Dixon (2002) discussed various summation methods of computing SDI, he only discussed the techniques used to compute them. The discussion here is to focus on the theoretical foundation of partitioning SDI and the applications of the theoretical foundation. The objective of this paper is to show (1) the theoretical foundation of Stage's SDI formulation, and (2) its application in partitioning SDI at the tree or group level.

STAND DENSITY INDEX

Reineke (1933) developed the following expression for stand density index (SDI) for even-aged stands:

$$\log(SDI) = \log(N) + k \cdot \log(D_q) - k = \log(N) + k \cdot \log(DD/N)^{1/2} - k \quad (1)$$

where

\log = the common logarithm function

$N = \sum_{i=1}^s EF_i$ = number of stems per acre

s = the number of stems in the plot

EF_i = the expansion factor, or the number of stems per acre represented by tree i

$D_q = (DD/N)^{1/2}$ = the quadratic mean diameter

$DD = \sum_{i=1}^s d_i^2 \cdot EF_i$ = the sum of squared DBH

d_i = DBH of the i^{th} stem

$k = 1.605$ is a constant.

Taking antilog of both sides of equation 1 yields:

$$SDI = N \cdot (DD/N)^{k/2} \cdot 10^{-k} = 10^{-k} \cdot N^{(1-k/2)} \cdot DD^{k/2} \quad (2)$$

Stage (1968) reformulated equation 2 as follows:

$$SDI = \sum_{i=1}^s (a + b \cdot d_i^2) \cdot EF_i = a \cdot N + b \cdot (\sum_{i=1}^s d_i^2 \cdot EF_i) \quad (3)$$

where

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$$a = 10^{-k} \cdot (1 - k/2) \cdot (DD/N)^{k/2} = 10^{-k} \cdot (1 - k/2) \cdot D_q^k = 0.004904 \cdot D_q^{1.605} \text{ and}$$

$$b = 10^{-k} \cdot (k/2) \cdot (DD/N)^{(k/2-1)} = 10^{-k} \cdot (k/2) \cdot D_q^{(k-2)} = 0.019927 \cdot D_q^{-0.395}.$$

Stage showed that the SDI computed from equation 2 is identical to that computed from the Reineke's original formulation. However, he did not mention the theoretical foundation that was used to derive equation 3. The theoretical foundation on partitioning SDI is presented.

PROPERTIES OF HOMOGENEOUS FUNCTIONS

Homogeneous Functions

According to Silberberg and Suen (2001), a real-valued function is called a homogeneous function of degree r if and only if the following relationship holds true:

$$f(t \cdot x_1, t \cdot x_2, \dots, t \cdot x_n) = t^r \cdot f(x_1, x_2, \dots, x_n) \quad (4)$$

where

t can be any value if $f(t \cdot x_1, t \cdot x_2, \dots, t \cdot x_n)$ lies within its domain. Homogeneity of degree 1 (i.e., $r = 1$) is a special case of homogeneous functions. It is also called linear homogeneity.

If a function is homogeneous of degree r , its first order partial derivatives are homogeneous of degree $r - 1$. In other words, its derivative with respect to t , i.e., $t \cdot t^{r-1} \cdot f(x_1, x_2, \dots, x_n)$, is also a homogeneous function.

Euler's Theorem

Euler's theorem (Silberberg and Suen 2001) states that a function is homogeneous of degree r if and only if the following relationship holds true:

$$r \cdot f(t \cdot x_1, t \cdot x_2, \dots, t \cdot x_n) = \frac{\partial f}{\partial x_1} \cdot x_1 + \frac{\partial f}{\partial x_2} \cdot x_2 + \dots + \frac{\partial f}{\partial x_n} \cdot x_n \quad (5)$$

$$r \cdot f(t \cdot x_1, t \cdot x_2, \dots, t \cdot x_n) = \frac{\partial f}{\partial x_1} \cdot x_1 + \frac{\partial f}{\partial x_2} \cdot x_2 + \dots + \frac{\partial f}{\partial x_n} \cdot x_n$$

Let r equal 1, equation 5 becomes:

$$f(t \cdot x_1, t \cdot x_2, \dots, t \cdot x_n) = \frac{\partial f}{\partial x_1} \cdot x_1 + \frac{\partial f}{\partial x_2} \cdot x_2 + \dots + \frac{\partial f}{\partial x_n} \cdot x_n \quad (6)$$

Example of Homogeneous Functions with Two Variables

For example, let $f(x,y)$ be a homogeneous function of degree r so that $f(t \cdot x, t \cdot y) = t^r \cdot f(x,y)$. Now define $u = t \cdot x$ and $v = t \cdot y$, so that $f(t \cdot x, t \cdot y) = f(u, v) = t^r \cdot f(x,y)$.

The first order derivative of $f(t \cdot x, t \cdot y)$ with respect to t is:

$$\frac{\partial f(t \cdot x, t \cdot y)}{\partial t} = \frac{\partial f}{\partial u} \cdot \frac{du}{dt} + \frac{\partial f}{\partial v} \cdot \frac{dv}{dt} = \frac{\partial f}{\partial (t \cdot x)} \cdot x + \frac{\partial f}{\partial (t \cdot y)} \cdot y \quad (7)$$

The first order derivative of $t^r \cdot f(x,y)$ with respect to t is as follows:

$$\frac{d[t^r \cdot f(x,y)]}{dt} = r \cdot t^{r-1} \cdot f(x,y) \quad (8)$$

Since $f(t \cdot x, t \cdot y) = t^r \cdot f(x,y)$ for a homogeneous function, the relationship is also true for their first order derivatives as follows:

$$r \cdot t^{r-1} \cdot f(x,y) = \frac{\partial f}{\partial (t \cdot x)} \cdot x + \frac{\partial f}{\partial (t \cdot y)} \cdot y \quad (9)$$

Let t in equation 9 equal 1, the result is:

$$r \cdot f(x,y) = \frac{\partial f}{\partial x} \cdot x + \frac{\partial f}{\partial y} \cdot y \quad (10)$$

Finally, when $r = 1$, we have:

$$f(x,y) = \frac{\partial f}{\partial x} \cdot x + \frac{\partial f}{\partial y} \cdot y \quad (11)$$

DERIVING STAGE'S SDI FORMULATION

Let $SDI = f(N, DD) = 10^{-k} \cdot N^{(1-k/2)} \cdot DD^{k/2}$ where $k = 1.605$. Multiplying a constant scalar t to N and DD yields:

$$f(t \cdot N, t \cdot DD) = 10^{-k} \cdot (t \cdot N)^{(1-k/2)} \cdot (t \cdot DD)^{k/2} = t \cdot 10^{-k} \cdot N^{(1-k/2)} \cdot DD^{k/2} = t \cdot f(N, DD)$$

By the definition of homogeneity, SDI is a homogeneous function of degree 1. Along with Euler's theorem, SDI becomes separable as follows:

$$SDI = \frac{\partial SDI}{\partial N} \cdot N + \frac{\partial SDI}{\partial DD} \cdot DD \quad (12)$$

where

$$\frac{\partial SDI}{\partial N} = 10^{-k} \cdot (1 - k/2) \cdot (DD/N)^{k/2} = 10^{-k} \cdot (1 - k/2) \cdot D_q^k = 0.004904 \cdot D_q^{1.605} = a \text{ and}$$

$$\frac{\partial SDI}{\partial DD} = 10^{-k} \cdot (k/2) \cdot (DD/N)^{(k/2-1)} = 10^{-k} \cdot (k/2) \cdot D_q^{(k-2)} = 0.019927 \cdot D_q^{-0.395} = b.$$

Equation 12 can be rewritten as follows:

$$SDI = a \cdot N + b \cdot DD = a \cdot N + b \cdot (\sum_{i=1}^s d_i^2 \cdot EF_i) \quad (13)$$

The coefficients a and b could be interpreted as the weight of the presence of a stem and the weight of the size of a stem, respectively.

PARTITIONING SDI

Tree Level

Let's define $SDI_i = (a + b \cdot d_i^2) \cdot EF_i$ as the contribution of stem i to the overall stand SDI. From equation 3, it is clear that

$$SDI = \sum_{i=1}^s SDI_i = \sum_{i=1}^s (a + b \cdot d_i^2) \cdot EF_i \quad (14)$$

Group Level

Suppose a stand is partitioned into m mutually exclusive groups. Let $SDI(j)$ be the SDI portion contributed by the j^{th} group, defined as follows:

$$SDI(j) = \sum_{i=1}^{g(j)} SDI_i = \sum_{i=1}^{g(j)} (a + b \cdot d_i^2) \cdot EF_i \quad (15)$$

where the summation sign extends to all stems $g(i)$ belonging to the j^{th} group, $j = 1, 2, \dots, m$.

Again, it is evident that

$$SDI = \sum_{j=1}^m SDI(j) = \sum_{i=1}^s (a + b \cdot d_i^2) \cdot EF_i \quad (16)$$

where $s = \sum_{j=1}^m g(j)$ and $g(j)$ is the number of stems in the j^{th} group.

Partitioning the SDI using the above methods ensures additivity not only at the tree level but also at the group level. Note that the contribution of a tree, either individually or as member of a group, is

not constant but depends on the coefficient of b , as well as the total number of stems in the stand.

DISCUSSION AND NUMERICAL EXAMPLES

Inventory data from four stands, provided by Washington State Department of Natural Resources (WADNR), were used to demonstrate the computation of SDI. These stands exhibit a wide range of diameter distributions (fig. 1).

Trees in each stand were divided into two groups based on DBH: (i) < 5 inches and (ii) ≥ 5 inches. Number of trees per acre (N), sum of squared diameters (DD), and quadratic mean diameter (D_q) were calculated separately for each group (table 1). Component SDI for each group was computed by use of Reineke's method (equation 2) and then Stage's formulation (equation 15).

Table 1 shows that Reineke's formula did not result in additivity: the sum of the two component SDIs did not equal the SDI computed from N and D_q for the stand as a whole. On the other hand, Stage's method resulted in component SDIs adding to total stand SDI. Additivity is attained with Stage's formulation, but not with Reineke's formula.

To illustrate the differential effects of stem numbers on SDI, 1,000 seedlings (< 4.5 feet in height and 0 inches in DBH) were added to N in the $DBH < 5$ inches group of each of the stand, shown in the highlighted rows of table 2; DD was unchanged, but D_q declined. Using Reineke's formula, changing stem numbers had relatively minor effects on either overall SDI or component SDI, comparing table 1 to table 2. Applying Stage's formulation, the additional stems reduced the value of coefficient a while increasing coefficient b for each stand. But the magnitude of change depended on the diameter distribution of stems and resulting D_q , for both overall SDI and component SDI. For a stand with a small D_q , increasing the number of stems had a relatively smaller effect on SDI than for a stand with large D_q . Alternatively, increasing the size of trees has a greater effect on stands with small D_q while increasing D_q had relatively less impact than increasing stem numbers for stands with large D_q .

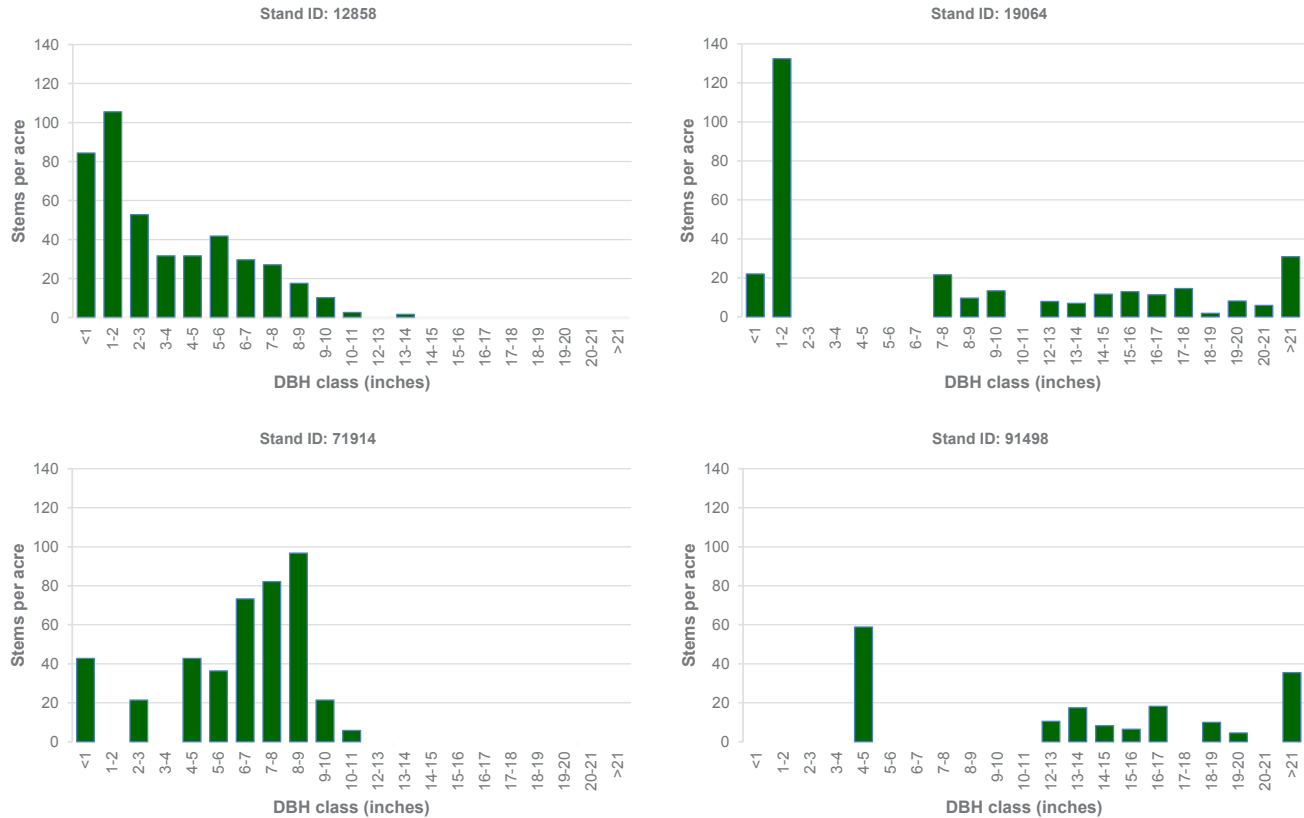


Figure 1—Diameter distribution of four different stands.

The contribution to stand density by a single stem varied from stand to stand. Because $SDI_i = (a + b \cdot d_i^2) \cdot EF_i$, even if two trees from different plots have the same diameter measurements, their contributions to the total SDI depend on the coefficients a and b , which in turn are functions of D_q . Assuming that plot size is 0.1 acre, the component SDI of a single tree of DBH 10 inches varies from 10.1 for stand 19064 to 12.5 for stand 91498, computed from values in table 1.

Stage's SDI formulation is useful for stand component groupings. It can be applied to groups of trees, based on specific DBH ranges, species (mixed-species stands), or age (uneven-aged stands). Contribution from each group to the total SDI can be readily computed from equation 15. Since Stage's SDI is based on individual trees, it can also be applied to multi-modal stands, similar to those depicted in figure 1. The component tree-level SDI can be considered as a competition index by group, which may be useful in analyzing within

stand and between tree-group growth and mortality variation.

In 1982, Curtis developed a density index called relative density (RD). It is available in some of FVS variants and is often used in place of SDI in the Pacific Northwest. $RD (= BA/\sqrt{D_q})$ can be expressed as a function of N and DD , i.e., $\frac{\pi}{24^2} \cdot N^{0.25} \cdot DD^{0.75}$. This is similar to Stage's formulation of SDI, except the constant term is $\pi/24^2$ instead of 10^{-k} and k is 1.5 instead of 1.605. By Euler's theorem, it can be partitioned as:

$$RD = \frac{\partial RD}{\partial N} \cdot N + \frac{\partial RD}{\partial DD} \cdot DD = a' \cdot N + b' \cdot (\sum_{i=1}^s d_i^2 \cdot EF_i) \quad (17)$$

where

$$a' = (0.25 \cdot \pi/24^2) \cdot (DD/N)^{0.75} = 0.001364 \cdot D_q^{1.5} \text{ and} \\ b' = (0.75 \cdot \pi/24^2) \cdot (DD/N)^{-0.25} = 0.004091 \cdot D_q^{-0.5}.$$

Table 1—Values of variables, by Stand ID

Variable	Stand ID			
	12858	19064	71914	91498
SDI (for entire stand)	108.5	427.4	224.7	556.9
Reineke (1933)				
Sum	96.8	375.9	218.8	520.2
DBH < 5"	21.3	3.5	14.7	13.5
DBH ≥ 5"	75.5	372.4	204.1	506.7
Stage (1968)				
Sum	108.5	427.4	224.7	556.9
DBH < 5"	27.5	42.9	19.7	43.8
DBH ≥ 5"	81.0	384.5	205.0	513.0
N				
Total	436.5	311.4	422.4	169.7
DBH < 5"	306.0	154.4	106.8	58.9
DBH ≥ 5"	130.5	156.9	315.6	110.8
DD				
Total	7,705.03	46,203.72	19,234.80	74,608.04
DBH < 5"	1,107.79	137.89	900.15	942.11
DBH ≥ 5"	6,597.24	46,065.83	18,334.65	73,665.93
D _q				
Total	4.20	12.18	6.75	20.97
DBH < 5"	1.90	0.95	2.90	4.00
DBH ≥ 5"	7.11	17.13	7.62	25.78
a	0.049105	0.271087	0.105047	0.648275
b	0.011303	0.007423	0.009374	0.005990

SDI=Stand density index; N=Number of trees per acre; DD=Sum of squared diameters; D_q=Quadratic mean diameter. The highlighted rows were the actual measurements of each stand.

RD can thus be used similarly to SDI and a number of existing FVS keywords and functions support its application, e.g., BRDen, ARDen, ThinRDen, and SpMcDBH. Since the use of ThinRDen or SpMcDBH can be for specific DBH ranges or species, partitioning RD may be advantageous.

CONCLUSION

Since the original Reineke's SDI formula possessed the properties of homogeneous functions, it guaranteed that Stage's formulation would work. By Euler's theorem, it ensured that the Stage's

SDI formulation is additive and produces the same results as the original SDI formula. The applications of the theoretical foundation include partitioning SDI both at the tree level and the group level. Such partitioning can be useful in targeting component growth and mortality. Evaluation of variation in the *a* and *b* coefficients could potentially extend the applications of SDI from even-aged stands to uneven-aged stands.

Table 2—Values of variables, by Stand ID

Variable	Stand ID			
	12858	19064	71914	91498
SDI (for entire stand)	137.3	567.7	285.6	815.4
Reineke (1933)				
Sum	103.9	377.6	227.4	530.7
DBH < 5"	28.4	5.2	23.3	23.9
DBH ≥ 5"	75.5	372.4	204.1	506.7
Stage (1968)				
Sum	137.3	567.7	285.6	815.4
DBH < 5"	40.5	100.1	54.6	154.0
DBH ≥ 5"	96.8	467.7	231.0	661.3
N				
Total	1,436.5	1,311.4	1,422.4	1,169.7
DBH < 5"	1,306.0	1,154.4	1,106.8	1,058.9
DBH ≥ 5"	130.5	156.9	315.6	110.8
DD				
Total	7,705.03	46,203.72	19,234.80	74,608.04
DBH < 5"	1,107.79	137.89	900.15	942.11
DBH ≥ 5"	6,597.24	46,065.83	18,334.65	73,665.93
D _q				
Total	2.32	5.94	3.68	7.99
DBH < 5"	0.92	0.35	0.90	0.94
DBH ≥ 5"	7.11	17.13	7.62	25.78
a	0.018878	0.085510	0.039650	0.137673
b	0.014301	0.009861	0.011914	0.008770

SDI=Stand density index; N=Number of trees per acre; DD=Sum of squared diameters; D_q=Quadratic mean diameter.

The highlighted rows were the numbers in table 1 plus 1,000 artificial seedlings (DBH=0") for each stand.

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Regeneration Modeling

Modeling the Impact of Overstory Density on the Regeneration Dynamics of Missouri Ozark Forests

Lance A. Vickers, David R. Larsen, Benjamin O. Knapp,
Daniel C. Dey, and John M. Kabrick¹

Foresters have long understood that overstory manipulation offers opportunities to influence regeneration dynamics. However, predicting the effects of silvicultural choices on the composition and structure of the regeneration layer remains difficult. Regeneration modeling is difficult for several reasons, one being that regeneration is an inherently complex and stochastic process, often with low signal:noise ratios and considerable uncertainty for statistical estimation. Another difficulty may be that the regeneration process, particularly in eastern deciduous forests, is often considered to span two distinct phases that Dey (2014) termed regeneration and recruitment. As a result, regeneration models in these forests have to account for both establishment and early stand dynamics to estimate the development and fate of reproduction. Finally, sufficient data for parameterizing regeneration models can be very expensive in cost, effort, time, and space, particularly if regional variations in species response to multiple and interacting factors are of interest. In summation, efforts to model regeneration can be plagued by theoretical, statistical, empirical, and economic insufficiencies. Given these hurdles, it is hardly surprising that an ideal framework for a full establishment model in the Forest Vegetation Simulator (FVS) for eastern deciduous forests has proven elusive despite the many advances in theory and practice that have resulted from modeling efforts in those forests.

To expand the regeneration modeling capacity in the deciduous forests of the Missouri Ozarks, we developed a collection of models that estimate establishment (Vickers and others 2017), allometry

(Vickers 2015), and growth (Vickers and others 2014) as a function of overstory density and other factors for several native species. While a pre-disturbance inventory is required, we strove to limit the required input to attributes that are either commonly inventoried or are otherwise relatively simple to collect. The covariates used in the models were predominantly based on stand development hypotheses, and empirical data were used for parameterization. Much of the data were collected as part of a long-term study (Shifley and Brookshire 2000), but our analyses largely focused on short-term responses using available early results.

There are three main sources of reproduction establishment in the mixed-hardwood forests of the Central Hardwood Region: (1) sprouting, (2) advance reproduction, and (3) new germination. To limit inventory demands, we suggest only potential sprouts and large advance reproduction ($ht \geq 1m$) be fully tallied and their establishment rates modeled via parameters gleaned from literature (e.g., Keyser and Loftis 2015, Knapp and others 2017, Vickers and others 2016). For some species, models are available to estimate the density of advance reproduction (Kabrick and others 2014, Larsen and others 1997). We modeled establishment from the more variable sources—small advance reproduction and new germination—using three pragmatic covariates: residual overstory, presence/absence of advance reproduction, and presence/absence of residual seed sources (Vickers and others 2017). This approach increases regeneration modeling efficacy by reducing the inventory effort required, focusing that effort on reliable sources of reproduction, and increasing compatibility for

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species not reliant on advance reproduction. Our models produce both deterministic and stochastic estimates of regeneration establishment and initial attributes shortly after disturbance (3 years). After that, techniques more common to growth and yield modeling, such as an annualized height growth models that incorporate initial height, residual overstory density, species, and site class, can be used to incrementally update the development of the cohort throughout the regeneration period (Vickers and others 2014, Vickers 2015).

Though further research and development are needed, the combination of these models provides a tool for both applied and empirical objectives and provides opportunities to increase both our understanding of the regeneration process and the efficacy of our efforts to manipulate it. To date, model estimates have shown that reproduction abundance and growth decreases with increasing residual overstory, provided evidence of interspecific differences in establishment and growth rates, and quantified how those differences vary with residual overstory density (Vickers and others 2014, 2017). These results are consistent with reports of regeneration response to various silvicultural manipulations in the Missouri Ozarks and beyond (Johnson and others 2009). The performance of our parameterized models outside the Missouri Ozarks is unknown and direct applications outside the region are not recommended. Nonetheless, we suspect that with refinement, the general approach and concepts used may be adaptable to other species and locales.

Despite the numerous hurdles involved, expanding the regeneration modeling capacity for eastern deciduous forests remains a clear need. Ideally, this expansion would anticipate the need for periodic updates and re-parameterization to accommodate changing conditions during development. Based on our recent modeling experiences, we suggest that successful expansion will require creativity and innovation. The call for creativity and innovation applies to the data requirements for parameterization and the efforts to collect it, the statistical techniques used for data analyses, and the

underlying ecological and silvicultural theory that synthesizes those analyses into a unified framework.

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Development and Assessment of Regeneration Imputation Models for National Forests in Oregon and Washington

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While regeneration is an essential component of stand development and can have a significant impact on the outcome of growth model projections, tools for automatically introducing natural regeneration are not always readily available in growth models (Weiskittel and others 2011). This is true for the Forest Vegetation Simulator (FVS) (Dixon 2002), which has the option of automatically including natural regeneration for just three geographic variants in the Northern Rocky Mountains (Ferguson and Carlson 1993, Ferguson and others 1986) and coastal Alaska (Ferguson and Johnson 1988).

Regeneration is a highly stochastic process and many of its driving factors are not easily documentable in stand inventory data (Weiskittel and others 2011). Estimation of regeneration is further complicated in areas with high structural complexity or high diversity of regenerating species (Oliver and Larson 1996, Ek and others 1997). In the Pacific Northwest, forests can range from monoculture stands to complex multi-cohort, multi-species stands. With only a user-specified approach available to managers using FVS in the Pacific Northwest, there is a need for natural regeneration models that can be easily incorporated and automated into FVS.

Imputation models were developed to estimate natural regeneration density and composition on National Forest System (NFS) lands in Oregon and Washington. The models were based on Forest Inventory and Analysis (FIA) and Pacific Northwest Regional Vegetation Monitoring data collected between 2004 and 2013. Regeneration was defined

as live trees with stems < 2.54 cm in diameter. A minimum height requirement of 0.15 m for conifers and 0.3 m for hardwood species was imposed to exclude first-year regeneration; in this sense, our analysis includes advanced reproduction, however for simplicity and consistency within the modeling community, we refer to this broadly as regeneration. Summary metrics were aggregated at the subplot-level to retain specific combinations of regeneration species and count data. Only naturally regenerated subplots were included in the sample. Regeneration for sprouting species was not modeled in this exercise. The region spans multiple climatic zones and vegetation types, and contains a total of 846 plant associations (Hall 1998). To account for this variability, models were based on broad Forest Plant Association Groups (FPAG), which were created by aggregating similar plant associations (see www.ecoshare.info for plant association guides).

All model development and analysis was conducted in R (R Development Core Team 2015), using the *yalmpute* package (Crookston and Finley 2008), a most similar neighbor-like imputation approach. Stand density index, basal area per acre (later converted to per hectare), and FVS-computed percent canopy cover were identified as predictor variables to define similarity between stands for nearest neighbor selection. To allow for a stochastic component and preserve naturally occurring species-count combinations, 10 nearest neighbor subplots were identified and one was selected at random. The regeneration species and densities from this subplot were then imputed to the target tree list.

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Separate models were developed for a total of 212 distinct FPAGs. Model performance results are presented for 59 FPAGs that contained adequate sample observations for validation ($n \geq 120$ subplots). To evaluate model performance, data for each model was separated into testing (25 percent) and training (75 percent) validation datasets. Given the random component to these models, 1000 imputation simulations were performed with the same testing and training dataset and validation statistics were averaged over the 1,000 simulations. Validation statistics of bias, mean absolute deviation (MAD), and root mean-squared error (RMSE) of regeneration stems per ha (TPH) were calculated using methods similar to those of Hassani and others (2004). Models were considered to have either low (< 1000), moderate (1000 – 2000), or high (> 2000) RMSE. Error rate (ER) in prediction was evaluated as the frequency with which a model incorrectly predicted the total presence or absence of regeneration. This was evaluated as Total ER if calculated as total regeneration regardless of species, and species-specific ER if calculated for an individual species within an FPAG. Models were considered to have either low (< 20 percent), moderate (20–50 percent), or high (> 50 percent) ER.

Over 80 percent of the 59 FPAG-specific models had low to moderate RMSE and all but one model had low to moderate Total ER Confidence intervals based on the 1000 simulations suggest some of the models consistently produced negative bias (overestimation), with fewer models tending towards consistent underestimation. Although RMSE, MAD, and bias did not appear to be substantially affected by sample size, RMSE and MAD tended to increase with increasing interquartile range of total seedling counts (regardless of species). The greatest species-specific ER contributions came from the most common species in the study area.

Predicting regeneration from an existing subplot using imputation allows for the inherent variability and ecological integrity of these FPAGs to be preserved. These models have the additional advantage of utilizing publically available data, and have the ability to easily incorporate new inventory data to improve model estimates. Despite restricting the models to predict regeneration

based only on common metrics that could be calculated from simple common stand exam plots (e.g. basal area per acre, SDI, and percent canopy cover), the FPAG-specific models performed relatively well. Further refinement of the FPAG classifications, as well as incorporating additional imputation predictor variables could improve model performance.

If a similar modeling approach were to be incorporated into a growth and yield model like FVS, these models have the added flexibility of allowing a user to specify whether higher or lower than average regeneration is expected at a site based on empirical knowledge. For example, if higher than normal regeneration is expected, the 10 nearest neighbor stands will be ordered based on total regeneration stems (regardless of species) and one of the top three plots with respect to greatest amount total regeneration will be randomly selected for imputation. In addition, using the FIA dataset allows for additional data to be incorporated annually.

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Evaluation of Base Model

Performance of FVS Variants in Relation to an Extensive Chronosequence and Remeasurement Dataset for Eastern White Pine (*Pinus strobus*, L.) in Central Maine

David Ray and Robert Seymour¹

Eastern white pine (*Pinus strobus*, L.) (EWP) has been referred to as the “tree that built America” (Germain and others 2016), and despite steady declines in extent and stocking, it remains one of the most important softwood timber species in the Northeastern United States. At present the commercial value of EWP is derived almost exclusively from sawtimber, which has important implications for stand density management, generally favoring low stocking and fast growth of individual trees (Seymour 2007). At the same time, EWP is long lived (ca 450 years) and capable of attaining very high biomass and carbon densities (D’Amato and others 2017), making it worthy of consideration for use in forest-based carbon mitigation projects—an objective that likely favors complete utilization of growing space and ecologically based rotations. Our objective is to use the Forest Vegetation Simulator (FVS) and related extensions to compare model predictions with observed stand development patterns for EWP. The initial step in this process, reported here, involves comparison of model predictions with stand dynamics documented from an extensive network of EWP plots. Further, we compare predictions obtained from the default Northeast Variant of FVS (NE) (Dixon and Keyser 2008) with those provided by the AcadianGY run script of the NE variant (hereafter, ACD) (Weiskittel and others 2012, 2017) available in FVSONline (Crookston and Shettles 2017).

Benchmarking data for this effort were provided by long-term research plots located in Central Maine representing a wide range of management, including: pre-commercial thinning (PCT) (n=3

plots, remeasured 3 times, aged 31-35 in 2016), conventional B-line thinning (n=8 plots, remeasured 4 times, age 68 in 2016), low-density thinning (n=8, remeasured 1-4 times, aged 45-68 in 2016), and a no-management chronosequence (control; n=14 plots, remeasured 1-5 times, aged 27-212 in 2016). Remeasurements ranged from 3 to 9 years, with an average interval of 5 years. At each visit permanently numbered trees were assessed as live or dead and measured for stem diameter (d.b.h.), total height, height to crown base, and assigned a crown class. All data were organized within an MS Access database formatted for use with FVS, and plots were run as stands. Model runs with NE version 1882 were carried out using the Suppose interface (Dixon 2002) whereas predictions associated with ACD v9.2 were obtained using FVSONline (Crookston and Shettles 2017). Due to unresolved issues with the mortality function for the ACD, this aspect was handled by the base model option available in FVSONline (Personal communication. 2017. Aaron Weiskittel, Associate Professor, University of Maine, 260A Nutting Hall, Orono, ME 04469). Other differences between the models that warrant consideration, include: (1) model calibration with observed growth data was only possible for NE, and (2) effects of site quality on tree growth are quantified differently, viz. NE uses site index (SI), whereas a climate site index (CSI) variable is employed by ACD.

Before carrying out simulations corresponding to characteristics of the EWP study-plots we implemented a series of long-term (200 years) “bare ground” simulations representing a range of planting densities (100 - 3,000 TPA) and site

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qualities (C/SI=40-60-80 feet) to compare model outputs with expected relationships (Cawrse and others 2009, Leary 1997). Results of that exercise, using the base model mortality function for ACD, suggested that both models generally conformed to expectations. Traditional SI was quantified using equations in Parresol and Vissage (1998); CSI exists as a spatial raster extending across the domain of the ACD model, containing species-invariant values for ~100-acre grid cells. The EWP study plots were distributed across five CSI grid cells having an average value of 44.8±2.9 feet and corresponding to values for measured SI of 66.2±4.2 feet. Contrary to expectation, no apparent relationship was observed between the two measures ($r = -0.037$, $p=0.925$), perhaps in part owing to the close proximity of the EWP study plots relative to the broad region across which CSI was mapped. Nevertheless, because these representations of site quality were expected to show correspondence, we carried out runs with ACD using both sets of values.

Working with our remeasurement data, runs conducted with NE yielded a more than four-fold difference in the large tree diameter growth multiplier for trees on the low-density plots (2.15) compared to those on the control plots (0.49); values for the PCT (1.45) and B-line (0.86) plots were intermediate. Evidently, growth modifiers representing stand density within the base model do not effectively translate into differential tree growth rates absent calibration, which may also be

related to stand density attributes of the data used to fit the model. Recognizing this stocking-related disparity, coupled with the inability of ACD to self-calibrate, we initially focused the between-model comparisons on predictions made relative to the self-thinning control plots. Basic stand parameters (trees per acre, TPA; basal area, BA; quadratic mean diameter, QMD) were evaluated in relation to two scenarios for each model: for NE we compared simulations which were either calibrated or not; for ACD we used the two measures of site quality, CSI and SI (fig. 1). Calibration resulted in across-the-board improvements for model predictions obtained from NE, whereas the use of measured SI reduced model Bias and RMSE of ACD only for QMD (table 1). Stand-level BA was best represented by the calibrated runs with NE, whereas observed TPA and QMD were more closely tracked by ACD using CSI as the measure of site quality. In all cases ($n=4$) model residuals were positively correlated for TPA and BA, but not QMD (fig. 1).

Next steps include using the EWP dataset to construct a SDI based maximum density line and adjust the self-thinning trajectory within NE, which should help improve trends in model residuals (i.e. TPA, fig. 1). Preliminary analysis of diameter growth projections for individual trees on the low-density plots suggests that the ability to use observed growth rates to calibrate NE confers a distinct advantage to the base model framework, in contrast to our findings for the control plots. More

Table 1—Performance statistics for stand level parameters (TPA, BA, QMD) associated with four simulations indexed below. Observed values are from 14 control plots from the Central Maine Eastern white pine dataset.

Statistic	Stem density (TPA)				Basal area (BA, square feet per acre)				Average stand diameter (QMD, inches)			
	^a NE	^b NE	^c ACD	^d ACD	^a NE	^b NE	^c ACD	^d ACD	^a NE	^b NE	^c ACD	^d ACD
Bias	183.52	48.57	27.22*	59.57	-12.53	0.13*	-0.35	-3.2	-0.59	0.34	0.09	0.04*
RMSE	1160.71	307.21	172.19*	376.78	79.22	0.79*	2.21	20.24	3.75	2.16	0.59	0.26*

TPA=Trees per acres; BA=Basal area; QMD=Quadratic mean diameter; NE=Northeast Variant of FVS; ACD=AcadianGy run script of the NE variant.

^aUncalibrated model run with NE.

^bCalibrated model run with NE.

^cACD, base mortality model, and Climate Site Index.

^dACD, base mortality model, and traditional Site Index.

* =Best performing model.

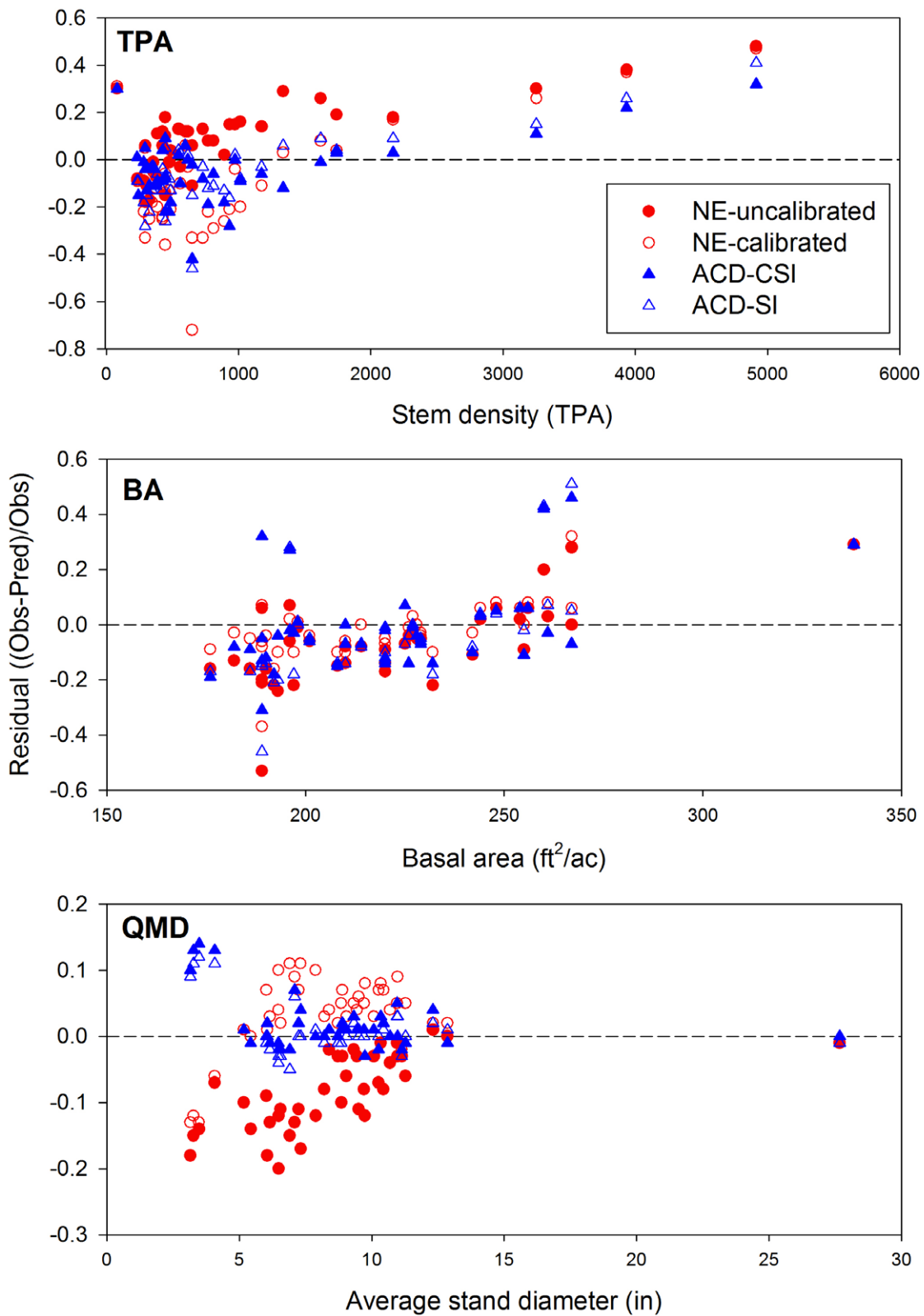


Figure 1—Distribution of relativized model residuals plotted across the range of associated stand level variables (TPA, BA, QMD) for control plots from the Central Maine Eastern white pine dataset. The four model runs correspond with those presented in table 1.

broadly, the dataset will be used to identify model adjustments necessary to produce reliable results for a range of density management scenarios, *sensu* Vandendriesche (2010).

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Using Forest Vegetation Simulator (FVS) to Calculate Cover Type Transition Probabilities of Deferred/Altered Stands Within the Border Lakes Subsection

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Abstract—Many Minnesota Department of Natural Resources (DNR) stands selected for examination have their timber sale deferred to a future period and many stands have their cover type “Altered” since it is felt the current cover type is incorrect. To better estimate harvest amounts during DNR planning efforts, it was decided to estimate how these altered/deferred stands may transition into other cover types. One method is the Forest Vegetation Simulator (FVS) Lake States (LS) growth and yield projection system. An Access database was downloaded from the FIA DataMart, FIADB version 5.1, Web site. The FIA2FVS translation tool was used to place FIA plots into FVS format. Plots were projected for 100 years with no management to see if transitions occurred by cover type. DNR field staff submitted “best guess” transition probabilities. These transition probabilities can be compared to transition projections from FVS. The field guesses are likely superior because they include local knowledge and better represent local growing conditions and are probably more applicable to those conditions that actually produce a deferral or alteration. It appears best to use probabilities provided by the DNR Areas, although FVS provides reasonable probabilities, there are some concerns, such as regeneration assumptions, inability to quantify factors that would better identify only those FIA plots with conditions similar to those stands that are likely to be deferred/altered, and substantial differences associated with the Balsam Fir [*Abies balsamea* (L.) Mill.] cover type. If FVS transition probabilities were to be used, these problems would need to be addressed.

INTRODUCTION

Many Minnesota Department of Natural Resources (DNR) stands selected for stand examination to have some type of management conducted have their timber sale deferred to a future period for a variety of reasons including low stocking, poor timber, etc., and many stands are coded as “Altered” during the current field/management inventories—hence, based on the current inventory, it is felt the cover type/forest type classification from the past is incorrect for these stands. To better estimate the amount of harvested timber on an annual basis during DNR Subsection Forest Resource Management Planning (SFRMP) efforts, it was decided to estimate the amount of annual volume not sold because of deferred/altered activities and to determine how these altered/deferred stands will transition over time, potentially into other cover types.

Several methods to estimate the likely transition of a cover type to another were examined and some were tried, however, an adequate methodology

was not identified. One potential method is to use the Forest Vegetation Simulator (FVS) Lake States (LS) growth and yield projection system (Dixon and Keyser 2008). FVS has the advantage that it can simulate a wide variety of forest types, stand structures, and species compositions.

The LS variant covers forest areas in the Great Lake states of Michigan, Minnesota, and Wisconsin. This includes Chippewa and Superior National Forests in Minnesota, Chequamegon and Nicolet National Forests in Wisconsin, and the Hiawatha, Ottawa, Huron and Manistee National Forests in Michigan.

METHODS

An Access database was downloaded from the Forest Inventory and Analysis (FIA) DataMart, FIADB version 5.1, Web site (<http://apps.fs.fed.us/fiadb-downloads/datamart.html>) for the State of Minnesota. The FIA2FVS translation tool (Vandendriesche 2014) was used to translate FIA plot data into an FVS formatted input database

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format readable by FVS. Within the FIA2FVS software, Plots only was selected (as opposed to projecting FIA subplots), and Inventory Year was used. Inventory Year is the year that best represents when plots were collectively scheduled to be measured while Measurement Year is the year in which the plot was actually sampled. Due to budgets, particularly in the Western United States, Inventory and Measurement Year can differ. The 272011 evaluation group (EVAL_GRP within FIADB) was selected, and all ownerships were included. For the 272011 evaluation group, Inventory and Measurement Year are essentially the same for all plots.

Only plots from the DNR Border Lakes subsection were included. This is land in northeastern Minnesota bordering Canada that roughly ranges from International Falls to the west and to Grand Marais to the south. Hence, transition probabilities should be more representative of cover types and forest conditions that would be observed within this subsection.

All ages were used to generate transition probabilities by cover type. Table 1 summarizes the original transition of FIA plots to another cover type (at some point within FVS there may be a second or third, etc., transition to an additional cover type). All stands inventoried during 2007, 2008, 2009, and 2010 were first projected to the common year of 2010 by FVS, LS variant, version 4862. This was the common point of determining the initial cover type. Several stands changed cover types during that 1 to 3 year period reflective of possible differences due to how the FIA cover typing algorithm is embedded in FVS. Projections were conducted for 100 years using a 10-year interval. No management within FVS was conducted, FVS was allowed to freely grow the stands.

Transition Probabilities from DNR Area Staff

In October of 2013 DNR Area field staff submitted “best guess” transition probabilities for seven jurisdictional management areas referred to as 117, 121, 221, 234, 245, 253, and 261 and are named Blackduck, Warroad, Deer River, Hibbing, Tower, Two Harbors, and Littlefork, respectively. Further summarizations were conducted by other DNR staff to develop the probabilities presented in table 1. These field transition probabilities can

be compared to those from FVS to verify FVS’ ability to predict transitions. Although the field guesses are not necessarily based on empirical data, they are likely superior because they include local knowledge and better represent local growing conditions, particularly given that FVS estimates are based on all ownerships to help increase sample size, not exclusively DNR lands. Additionally, the field guesses are probably more applicable to those conditions that actually produce a deferral or alteration. The Border Lakes subsection only includes Areas 245, 253, and 261 from above, also parts of Area 241 – Orr are included. Boundaries of subsections are based more on vegetational characteristics and therefore often include only parts of the jurisdictional Areas.

RESULTS

For ABg, aspen and balsam poplar (*Populus* spp.), there is general agreement among the transition of cover types, but the probabilities differ substantially. Based on both the subjective and empirical methods, it appears that transitions to oak (*Quercus* spp.), white pine (*Pinus strobus* L.), red pine (*Pinus resinosa* Ait.), and white spruce (*Picea glauca* (Moench) Voss) are rare. Additionally, both methods seem to show that transitions to northern hardwoods are relatively common (e.g., 24.1 percent and 16.2 percent). The DNR personnel predict most transitions occur to lowland hardwoods and birch (*Betula papyrifera* Marsh.) stands, while FVS projects most stands transition to balsam fir [*Abies balsamea* (L.) Mill.]. The different probabilities are substantial (e.g., 46.3 percent versus 2.7 percent for balsam fir).

For birch, there is a strong agreement among the transition of cover types, but similar to the ABg cover type transitions, the probabilities differ substantially in most cases. Based on both methods, it appears that transitions to lowland hardwoods, oak, white pine, and red pine are rare (as well as white spruce which doesn’t occur for either approach). Additionally, similar to the ABg cover type transitions, both methods seem to show that transitions to northern hardwoods are relatively common (e.g., 22.2 percent and 26.6 percent). The DNR personnel predict most birch cover type transitions occur to ABg, while FVS projects most transition to balsam fir. For the birch cover type, the

Table 1—Cover types transitioned from the original cover type to a new cover type

Old cover type	New cover type	Number	Percent of total	Minimum age	Area percent of total	
9 (LH & Ash)	12/14	ABg	4	44.4	32	-
	13	Birch	1	11.1	11	-
	62	BF	3	33.3	80	-
	71	BSL	1	11.1	10	-
12/14 (ABg)	9	LH & Ash	3	5.6	53	29.7
	13	Birch	3	5.6	29	18.9
	20	NH	13	24.1	17	16.2
	30	Oak	1	1.9	87	-
	51	EWP	2	3.7	95	-
	52	RP	2	3.7	55	-
	53	JP	1	1.9	36	8.1
	61	WS	2	3.7	34	-
	62	BF	25	46.3	11	2.7
	71	BSL	2	3.7	36	16.2
13 (Birch)	73	NWC	-	-	-	8.1
	9	LH & Ash	1	5.6	82	-
	12/14	ABg	2	11.1	39	51.6
	20	NH	4	22.2	54	26.6
	30	Oak	-	-	-	3.1
	51	EWP	1	5.6	87	-
	52	RP	-	-	-	3.1
20 (NH)	62	BF	10	55.6	25	15.6
	9	LH & Ash	1	14.3	63	-
	12/14	ABg	1	14.3	14	-
	13	Birch	1	14.3	82	-
	62	BF	3	42.9	43	-
	71	BSL	1	14.3	32	-
51 (EWP)	62	BF	2	100.0	27	-
52 (RP)	12/14	ABg	2	28.6	19	32.7
	13	Birch	2	28.6	82	28.8
	51	EWP	1	14.3	143	-
	53	JP	1	14.3	35	25.0
	61	WS	-	-	-	13.5
	62	BF	1	14.3	93	-
53 (JP)	12/14	ABg	1	7.7	26	-
	51	EWP	1	7.7	115	-
	52	RP	2	15.4	26	-
	61	WS	1	7.7	126	-
	62	BF	8	61.5	53	-
61 (WS)	71	BSL	1	100.0	71	-
62 (BF)	9	LH & Ash	-	-	-	5.5
	12/14	ABg	1	33.3	72	25.5
	13	Birch	-	-	-	27.3
	20	NH	1	33.3	156	5.5
	51	EWP	-	-	-	5.5
	61	WS	1	33.3	52	-
	71	BSL	-	-	-	12.7
	73	NWC	-	-	-	18.2
71 (BSL)	9	LH & Ash	-	-	-	6.4
	12/14	ABg	2	8.7	60	19.1
	13	Birch	1	4.3	162	4.3
	20	NH	1	4.3	143	-
	51	EWP	2	8.7	71	-
	61	WS	2	8.7	49	6.4
	62	BF	15	65.2	62	10.6
	72	Tamarack	-	-	-	31.9
	73	NWC	-	-	-	17.0
	74 (BSU)	BSU	-	-	-	4.3
72 (Tamarack)	51	EWP	1	20.0	65	-
	62	BF	1	20.0	102	-
	71	BSL	3	60.0	40	-
73 (NWC)	13	Birch	1	16.7	143	-
	62	BF	5	83.3	30	-

For cover types, numbers are numerical codes used by the DNR (e.g., 13 refers to a birch cover type). The category number is the number of FIA plots transitioning from the old cover type to the new cover type. Percent of total is the percent transition of all FIA plots that had a transition (not of all FIA plots) to a particular new cover type. Minimum age is the youngest age where a transition occurred to a new cover type. Also shown are estimates from the DNR Areas (area percent of total); for some old cover types, transitions occurred for either FIA or from the DNR areas that did not occur in the other estimation procedure.

LH—Lowland hardwood; ABg—Aspen/Balm; NH—Northern hardwoods; EWP—Eastern white pine; RP—Red pine natural; JP—Jack pine; WS—White spruce natural; BF—Balsam fir; BSL—Black spruce lowland; BSU—Black spruce upland; NWC—Northern white cedar.

different transition probabilities are substantial for ABg and balsam fir (e.g., 11.1 percent versus 51.6 percent for ABg).

For red pine, there is a strong agreement among the transition of cover types. Probabilities of transitions appear to be more agreeable as a whole compared to the ABg and birch cover type transitions, but some differ substantially. Based on both methods, it appears that transitions to black spruce [*Picea mariana* (Mill.) B.S.P.], lowland hardwoods, oak, and northern hardwoods are rare. Both methods seem to show that transitions of red pine to ABg, birch, and jack pine cover types are common. However, the methods differ to some degree on red pine transitions to white pine, white spruce, and balsam fir, but the differences in probabilities are lower in magnitude than some of the transitions observed for the ABg and birch cover types.

For balsam fir, there appears to be weak agreement among the transition of cover types, with the exception of the ABg cover type – both methods have high probabilities. Based on both methods, it appears that transitions to lowland hardwoods and white pine are rare. However, the methods differ substantially on the importance of transitions to white spruce, northern hardwood, birch, black spruce lowland, and northern white-cedar (*Thuja occidentalis* L.) cover types.

For black spruce, there appears to be weak agreement among transitions to other cover types, with the exception of the ABg, birch, and white spruce cover types. Based on both methods, it appears that transitions to lowland hardwoods, northern hardwoods, white pine, and black spruce upland are rare. However, the methods differ substantially on the importance of transitions to balsam fir, tamarack (*Larix laricina* (Du Roi) K. Koch), and northern white cedar cover types (once again substantial differences in transition to balsam fir – 65.2 percent versus 10.6 percent).

DISCUSSION

These FVS simulations may be viewed as out-of-the-box simulations where no attempt to adjust default parameters was undertaken. One potential problem associated with this FVS methodology is that the analysis relied on background and

density related mortality estimates and did not include disturbance mortality in the simulations. Disturbance agents such as Eastern Larch Beetle (*Dendroctonus simplex* LeConte, ELB), Emerald Ash Borer (*Agrilus planipennis* Fairmaire, EAB), and others such as birch decline, Spruce Budworm [*Choristoneura fumiferana* (Clemens)], etc., could impact transition probabilities between the cover types. The inability to account for these agents/factors when currently utilizing FVS may result in different transitions from cover types to others if indeed they were accounted for. For example, would a northern hardwood stand truly transition to a birch stand today at age 82 (see table 1) given Birch Decline, etc., as compared to probabilities from the past?

There seems to be a fair amount of discrepancy in the transition probabilities to balsam fir cover type between FVS and the DNR personnel that could be due to differences in the interpretation (e.g., algorithms in FVS) of what constitutes a balsam fir cover type. Differences in cover type interpretation could be a problem for any of the cover types, but balsam fir seems to have the most discrepancies.

To increase sample size, projections from all stand ages when using FVS were included. This could create discrepancies among FVS and the DNR personnel because in general the DNR personnel are likely basing their probabilities on more mature stands and stands that are to be harvested in the near future (although not exclusively). Additionally, discrepancies may occur because DNR personnel probabilities are likely based on those stand conditions more likely to actually generate deferrals/alterations—it is somewhat difficult to select FIA plots based on these “real world” conditions because it is somewhat difficult to actually express/quantify these conditions.

A potentially large caveat associated with FVS is its ability to model regeneration. FVS has a very simple regeneration assumption as a default. Alternatively, a user can supply their own probabilities of regeneration which would likely result in some different transitions than those reported in table 1. Transition probabilities presented in table 1 are based on the simplistic assumption of no natural regeneration over time. Future work would need to concentrate on including

empirical regeneration estimates into the FVS projections.

The age of stands within FVS may not always be a good indicator of stand development and structure because in fact most of these stands are likely uneven-aged—within DNR SFRMP analyses similar problems exist because uneven-aged stands are essentially modeled as if even-aged. Thus, ages of stands within FVS and transition ages should probably be viewed in light that they are more indicative of time passed rather than stand ages that are representative of stages of stand development and structure. For instance, there is a transition from a Balsam Fir cover type to a northern hardwood stand when the fir stand is 156 years old—most likely there are no even-aged balsam fir stands of this age.

CONCLUSIONS

At the current time, it appears best to use probabilities provided by the DNR personnel. Although FVS provides reasonable probabilities, there are some issues related to using FVS including the regeneration assumptions, inability to quantify factors that would better identify only those FIA

plots with conditions similar to those stands that are likely to be deferred/alterd, and substantial differences associated with the Balsam Fir cover type. If FVS is used, these problems should be addressed. The DNR is currently developing a program to monitor and track what stands are altered and deferred. This empirical information can be used in planning efforts to better estimate how these stands may transition to other cover types.

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Evaluating Diameter Increment in Disturbed Forests Across the U.S. Lake States

Macklin Glasby and Matthew Russell¹

Due to the recent climatic and weather pattern changes, improving growth and yield models to better account for disturbances and stochastic events is now crucial for managers to successfully manage forests under uncertainty. Current diameter increment equations are not sensitive to forest disturbance which may accelerate or decelerate individual tree growth. There is a need to accurately represent disturbance agents so that forest managers can implement silvicultural strategies in an attempt to reduce the forest health impacts caused by disturbance (Fox and others 2001, Russell and others 2015).

Using 15 years of diameter increment observations from three measurements of Forest Inventory and Analysis (FIA) plots across the U.S. Lake States (Michigan, Minnesota, and Wisconsin), we assessed the performance of recently developed diameter increment equations in disturbed and non-disturbed forests. Disturbances in the FIA plots were categorized using broad disturbance classes to ensure adequate sample sizes and included damages due to animal, disease, fire, insect, and weather (in addition to no disturbance). If a plot record contained a disturbance code, it indicated the plot experienced a disturbance since the last plot inventory (i.e., within the last 5 years). Approximately 6 percent of all FIA plots inventoried between 1999 and 2014 experienced one of the most common forest disturbances ($n = 2,694$). Animal and weather were the most common forest disturbances, accounting for 1.7 percent and 1.4 percent of all FIA observations, respectively.

Diameter increment equations recently developed by Deo and Froese (2013) for the Lake States and Central States were used in this analysis. Species-specific equations used tree size and vigor,

competition, and site quality variables in a single equation with an intercept and up to 10 covariates. For 22 species analyzed, these diameter increment equations overpredicted 10-year diameter increment slightly (mean bias of 0.03 inches/10-years) in forests that did not experience a disturbance. In disturbed forests, mean bias of 10-year diameter growth of disturbed trees was 0.12 inches/10-years, indicating that equations underpredicted diameter increment in disturbed forests. When analyzed by species, the Deo and Froese (2013) predictions displayed the largest mean bias for hardwoods commonly found in the region. American elm (*Ulmus americana* L.) had the highest mean bias for both disturbed and non-disturbed trees, averaging underpredictions of 0.94 ± 1.34 and 0.71 ± 1.17 in/10-years, respectively. Quaking aspen (*Populus tremuloides* Michx.), a dominant species in the region, performed poorly on disturbed plots with a mean bias of 0.58 ± 0.94 in/10-years (table 1).

While diameter increment equations performed well on average, it is apparent that when applied to individual species, the predictions will underestimate or overestimate diameter increment. Accounting for biotic disturbance agents (e.g., insects and diseases) is extremely important when trying to generate realistic predictions of stand level growth (Woods and Coates 2013).

This analysis could help modelers to improve the performance of growth and yield models in the presence of disturbance and better quantify the uncertainty of forest growth following disturbance. Similarly, this benchmarking exercise is important to identify future improvements to growth and yield models such as the Forest Vegetation Simulator when used in the U.S. Lake States.

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Table 1—Average observed and predicted 10-year diameter increment (ΔDBH_{10}) for the three most common conifer and hardwood species for Forest Inventory and Analysis data collected between 1999 and 2014

Species	Disturbed?	n	Mean observed ΔDBH_{10}	Mean predicted ΔDBH_{10}
<i>inches per 10 years</i>				
Conifers				
<i>Abies balsamea</i>	No	15,906	1.047	0.947
	Yes	386	1.312	0.958
<i>Picea mariana</i>	No	16,273	0.546	0.675
	Yes	287	0.676	0.765
<i>Pinus resinosa</i>	No	14,681	1.263	1.012
	Yes	91	1.479	1.039
Hardwoods				
<i>Acer saccharum</i>	No	32,592	0.687	0.898
	Yes	1102	0.776	0.880
<i>Populus tremuloides</i>	No	28,241	1.415	1.040
	Yes	993	1.626	0.914
<i>Fraxinus nigra</i>	No	13,862	0.636	0.859
	Yes	340	0.696	0.868

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Evaluation of the FVS-CR Diameter Growth Model and Potential Modifications in Structurally-Complex Ponderosa Pine Forests

Yvette L. Dickinson, Michael A. Battaglia, and Lance A. Asherin¹

Forest managers are increasingly focused on increasing or maintaining forest spatial complexity; however, many of the forest growth and yield models commonly used by forest managers assume that stands are structurally homogeneous. Using these homogeneous models to predict the development of spatially complex stands through time may provide inaccurate results. Forest managers require accurate predictions of future stand structures under a range of conditions to understand the impact of silvicultural treatments on various ecosystem services and inform the planning of silviculture prescriptions. This is particularly concerning in the dry forests of the Western United States because forest managers are currently implementing silvicultural treatments to restore the heterogeneous, multi-aged, open-woodland structure that was typical prior to European settlement (Churchill and others 2013), but may not be able to accurately predict the outcomes of these restoration treatments over time.

In silviculture treatments that aimed to increase forest structural complexity, we investigated the accuracy and precision of stand and individual tree diameter growth estimates made by Forest Vegetation Simulator - Central Rockies Variant (CR, version 1305) in six adjacent 4-ha ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) stands in the North Kaibab Plateau, AZ, over a 16-year period. CR is a non-spatial, individual tree growth model based on the GENGYM model (Edminster and others 1991, Keyser and Dixon 2008), and is commonly used by forest managers to understand likely future stand conditions. The treatments included two untreated controls and four group-

selection harvests using a q-ratio of 1.1 for 2.54 cm size classes to leave a total residual basal area of 14.3 m² ha⁻¹ (low residual) and 16.8 m² ha⁻¹ (high residual), respectively.

All trees in the stands were measured and mapped in 1994 following harvest, and again in 2001 and 2010. In total 8,503 trees were measured and mapped. Trees within 24 m of the edge of the plot were determined to be edge trees, and were excluded from the analysis as focal trees. The remaining trees were randomly divided into a training dataset of 3,475 trees, and a testing dataset of 160 trees.

To investigate the accuracy of the current CR diameter growth model, the measured diameter growth over the 16-year measurement period was contrasted with the individual tree diameter growth predicted by the original model. In addition, to investigate the potential incorporation of spatial complexity into the model we examined whether the addition of 28 tree vigor, semi-distance independent and spatially explicit indices of local competition to the model improved the accuracy and precision of the estimates. The correlation coefficient (R²), bias, root mean square error (RMSE) and Bayesian information criterion (BIC) were calculated for each investigated model (table 1). The 28 indices investigated included the addition of local basal area, sum of local DBH, local sum of tree height (each including and excluding the focal tree), local tree density, Hegyi index of competition, and a modified Hegyi index based on tree height within 6, 12, and 24 m neighborhoods, and tree crown ratio.

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Table 1—The coefficient P-value, correlation coefficient (R^2), bias, root mean square error (RMSE) and Bayesian information criterion (BIC) for 29 assessed models, based on data from the testing dataset (160 trees randomly selected and retained from the dataset)

Model	Neighborhood size (m)	Coefficient P-value	R^2	Bias (cm decade ⁻¹)	RMSE (cm decade ⁻¹)	BIC
Original	N/A	N/A	0.9845	0.36	1.46	9849
Crown ratio	N/A	<0.0001*	0.9954	-0.15	1.39	1184
Local BA excluding the focal tree	6	<0.0001*	0.9946	0.45	1.46	9747
Local BA excluding the focal tree	12	0.4735	0.9946	0.38	1.46	9671
Local BA excluding the focal tree	24	0.0027*	0.9947	0.29	1.45	9416
Local BA	6	0.0006*	0.9946	0.44	1.46	9754
Local BA	12	0.6379	0.9946	0.38	1.46	9671
Local BA	24	0.0024*	0.9947	0.29	1.45	9415
Local sum of DBH excluding the focal tree	6	<0.0001*	0.9946	0.50	1.46	9721
Local sum of DBH excluding the focal tree	12	0.1075	0.9946	0.40	1.46	9669
Local sum of DBH excluding the focal tree	24	0.0014*	0.9947	0.29	1.45	9414
Local sum of DBH	6	<0.0001*	0.9946	0.49	1.46	9732
Local sum of DBH	12	0.1309	0.9946	0.40	1.46	9669
Local sum of DBH	24	0.0016*	0.9947	0.29	1.45	9415
Local sum of tree height excluding the focal tree	6	<0.0001*	0.9947	0.51	1.45	9714
Local sum of tree height excluding the focal tree	12	0.0075*	0.9946	0.43	1.46	9664
Local sum of tree height excluding the focal tree	24	0.0218*	0.9947	0.31	1.46	9419
Local sum of tree height	6	<0.0001*	0.9946	0.50	1.46	9724
Local sum of tree height	12	0.0100*	0.9946	0.42	1.46	9665
Local sum of tree height	24	0.0237*	0.9947	0.31	1.46	9419
Local tree density	6	0.9532	0.9946	0.37	1.46	9766
Local tree density	12	0.1133	0.9946	0.33	1.46	9669
Local tree density	24	0.0006*	0.9947	0.28	1.45	9413
Hegyí index of competition	6	0.0051*	0.9947	0.39	1.46	9849
Hegyí index of competition	12	0.6530	0.9947	0.37	1.46	9857
Hegyí index of competition	24	0.0045*	0.9947	0.33	1.46	9849
Modified Hegyí index based on tree height	6	0.0214*	0.9947	0.39	1.46	9852
Modified Hegyí index based on tree height	12	0.8265	0.9947	0.36	1.46	9857
Modified Hegyí index based on tree height	24	<0.0001*	0.9947	0.28	1.45	9835

*Statistically significant at alpha = 0.05.

N/A = Not applicable.

We found that while the stand-scale estimates of growth from the original model were relatively accurate with small errors in basal area growth estimates ($<1 \text{ m}^2 \text{ ha}^{-1} \text{ decade}^{-1}$), the estimates of individual tree diameter growth had a RMSE $1.46 \text{ cm decade}^{-1}$ (table 1), which equates to 44 percent of the mean individual tree diameter growth over a decade. Further, consistent with Ex and Smith (2014) and Petrova and others (2014), we found there is greater bias in the growth of small than large trees ($0.55 \text{ cm decade}^{-1}$ for trees $< 25.9 \text{ cm DBH}$, compared with $0.29 \text{ cm decade}^{-1}$ for trees $> 25.9 \text{ cm DBH}$), which is likely to produce inaccurate estimates of structural development over time. This variation in model bias suggests that small trees within the stand will grow even slower relative to large trees than predicted by the model.

The results of this study corroborate other studies that indicated crown ratio is an important predictor of individual tree growth (Ex and Smith 2014, Wykoff 1990). However, it is surprising that the addition of crown ratio to the model reduced the bias and RMSE of the projected individual tree diameter growth more than the addition of semi-distance independent and spatially explicit indices (table 1). The addition of crown ratio halved model bias to $-0.15 \text{ cm decade}^{-1}$ and reducing the RMSE to $1.39 \text{ cm decade}^{-1}$. Furthermore, there was no difference in the bias of small and large trees found for this revised model. Therefore, we recommend that crown ratio be added to the model to improve estimates of individual tree diameter growth in spatially complex stands, and improve estimates of stand structural development over time.

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Evaluation of the Fire and Fuels Extension

Evaluation of the Fire and Fuels Extension to the Forest Vegetation Simulator Within the Missouri Ozarks

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The Forest Vegetation Simulator (FVS) is a stand-based, individual-tree growth and yield model designed and maintained by the U.S. Department of Agriculture Forest Service. It is used by land managers on public and private ownerships to simulate and compare the effects of silvicultural treatments on forest stand dynamics including tree growth, mortality, and regeneration. The Fire and Fuels Extension (FFE) is an additional package of models designed to extend FVS to model fire effects on forest development and changes in fuel loading through time. Thus, it allows users to simulate fuel reduction treatments. FVS-FFE is a widely available model that links fuel dynamics and fire behavior predictions to stand-scale models of forest growth and yield. This link can increase model functionality by quantifying dynamic interactions among forest growth, mortality, fuels and fire. For example, wildfire risk is partly dependent on canopy fuels, which are modeled as a function of growth in FFE-FVS.

Originally released in 1997, FFE-FVS was created by combining existing fire models with the overstory growth models of FVS (Rebain 2010, Teck 1997). Included in the extension are Rothermel's fire spread model (Rothermel 1972), with two sets of fuel models commonly used in combination with Rothermel's model to calculate fire spread (Anderson 1982, Scott and Burgen 2005), and the First Order Fire Effects Model (FOFEM) (Reinhardt and Crookston 2003). This allows FFE-FVS to model both the background processes involved with forest fuels including accumulation and decomposition, as well as

conditional processes such as consumption when a fire is simulated.

The underlying Rothermel and FOFEM models have been developed independently of FFE-FVS and each other, often using data from areas of the United States different from where the models are subsequently applied. FFE-FVS is parameterized at the variant level, and uses default values to assign a value to each stand at the beginning of simulations if a user does not supply local parameter values. The default values are derived from Forest Inventory and Analysis (FIA) data collected across the geographic extent of the variant, which may encompass broad ecological variation. Without validation, managers cannot determine if model output from the FFE-FVS is realistic or what the effective geographical or ecological limits of the model are. Understanding the limits of FFE-FVS would result in better decision-making efforts by managers.

We tested whether the default model fuel loading values from two FVS variants (Central States (CS) and Southern (SN) variants, version 1860) were representative of field-based fuel load estimates from FIA data collected in the Ozark Highland region (Dixon and Keyser 2008, Keyser 2008). Data were collected in the down woody debris sampling, as part of the Phase 3 (P3) sampling design of FIA, and included 1 hour, 10 hour, 100 hour, and 1,000 hour fuel classes, as well as litter and duff loading. We used the FIA data to construct a bootstrapped distribution of mean fuel loading for six fuel categories and two forest-type groups (pine-oak

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and oak-hickory). We then compared the FFE-FVS default values against the distributions and tested for equivalency (i.e, do FFE-FVS values overlap the 95 percent confidence interval of the FIA data).

We also compared fuel loads projected through time by FFE-FVS to empirical data collected from a 14-year study in the Missouri Ozarks. Differences in fuel loading between projected and observed were calculated by fuel category and tracked through time across four different treatments (harvest, prescribed burn, harvest + burn, and control) using the CS and SN variants. The fire submodels of FFE-FVS use coarse categories as an input parameter rather than the explicitly projected values, so FFE-FVS will translate the projected values into one of the fuel categories prior to initializing a burn simulation. To test the efficacy of this technique, we compared the proportion of stands correctly classified at initialization and at the end of the projection.

FFE-FVS default surface fuel loading values were found to be largely unrepresentative of measured fuel loading in the Ozark region (table 1). For the oak-hickory forest type group (FIA numeric code 500), CS default values were included in the

confidence interval for the 1 hour, 100 hour, and litter classes, while SN default values were included for the 1 hour and 1,000 hour classes. In the oak-pine forest type group (FIA numeric code 400), CS default values were not included in any of the classes while SN default values were included in the 1,000 hour and litter class. The only situation where both variant values were inside the bounds of the confidence interval for the observed FIA data was for 1 hour fuels in the oak-hickory group. The default values were found to be directionally similar, as there were no cases where one variant was higher than the observed range while the other was below. The values were consistently both greater than the observed range with the exception of the litter class in the oak-hickory forest-type. The SN value was within the range of the confidence interval while the CS value was lower than the interval.

Results suggest that choice of variant did not significantly change projected fuel loading for all fuel classes at the end of the 14-year simulation (fig. 1), but differences in overstory models may cause longer term projections to diverge. The use of observed fuel values rather than defaults can improve projection accuracy in the short-term,

Table 1—FFE-FVS default values and bootstrapped confidence intervals for mean fuel loading (tons per acre) by category and forest-type group

Forest type group	Fuel hour class	2.5 percentile	Distribution mean	97.5 percentile	Central States default value	Southern default value
Oak-hickory	1 hour	0.090	0.118	0.173	0.150	0.130
Oak-hickory	10 hour	0.468	0.531	0.600	0.740	0.680
Oak-hickory	100 hour	1.415	1.636	1.873	1.700	1.930
Oak-hickory	1,000 hour	1.947	2.711	3.644	4.030	2.450
Oak-hickory	Litter	5.072	5.515	5.972	5.170	4.280
Oak-hickory	Duff	1.352	1.616	1.921	4.520	5.910
Oak-pine	1 hour	0.066	0.100	0.135	0.180	0.180
Oak-pine	10 hour	0.257	0.389	0.525	0.750	0.770
Oak-pine	100 hour	0.784	1.368	2.040	2.420	2.170
Oak-pine	1,000 hour	0.557	1.193	1.989	2.600	1.950
Oak-pine	Litter	2.713	3.833	4.929	5.370	4.070
Oak-pine	Duff	0.926	1.715	2.722	3.070	6.150

Bolded values are included in the confidence interval.

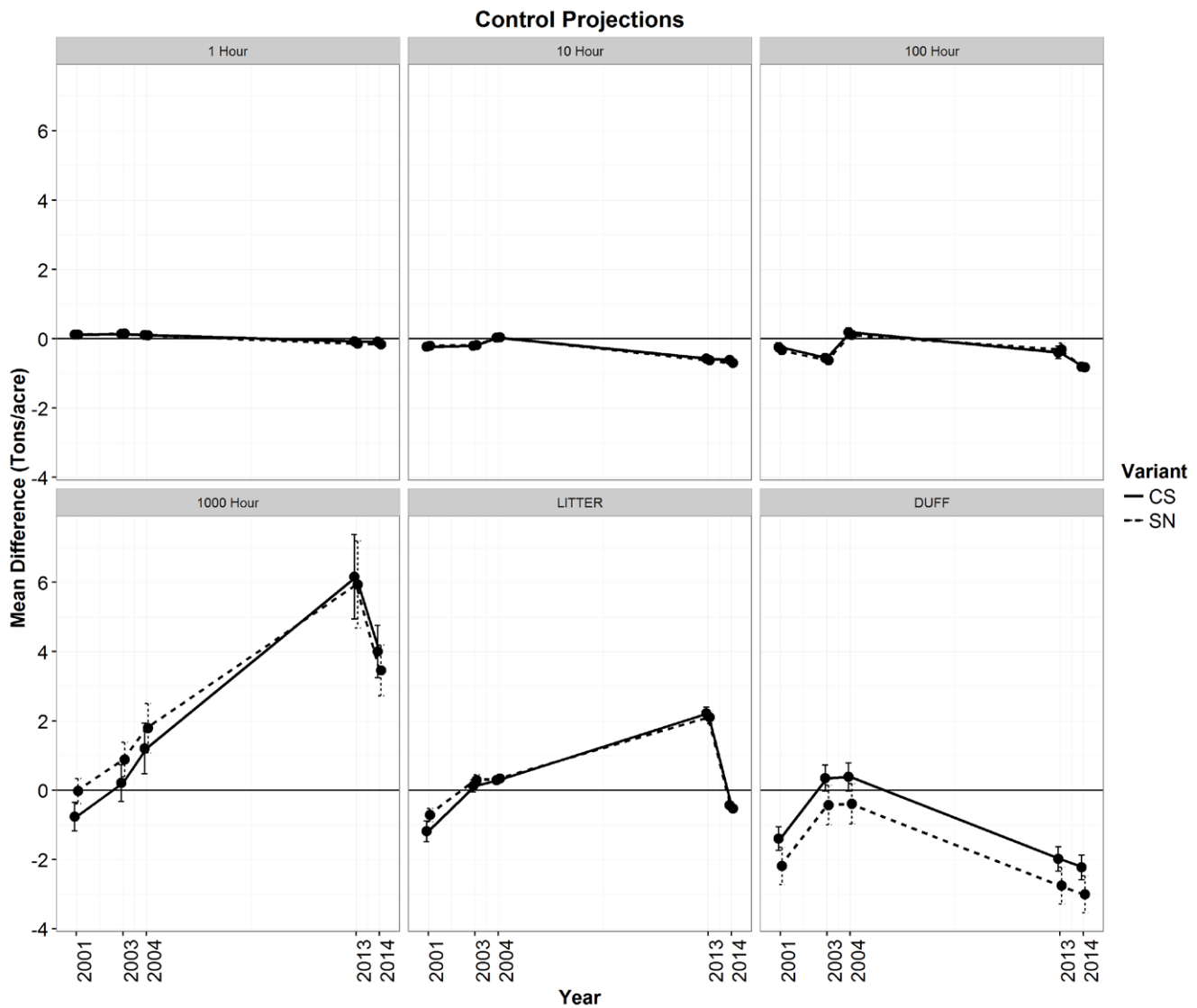


Figure 1—Mean difference between projected fuel values and observed fuel values for the control plots by variant for six fuel categories. Horizontal black line is at 0.

but longer term projections converged. When we classified field-based fuel loads into the fuel models used by FVS-FFE, we found that significant differences between study treatments in the accuracy of FVS-FFE classification, with harvest and control being classified incorrectly (42 percent and 53 percent) more than the harvest/burn and burn only (56 percent and 66 percent). These results suggest that FVS-FFE is a suitable tool for use by managers in the Central States region when they are planning prescribed fire operations. The internal model logic of aggregating projected fuel values into coarse fuel models can buffer error. Projecting

fuel loads after harvests or no treatment scenarios may require additional model calibration to achieve realistic projections.

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Validation and Development of Postfire Mortality Models for Upland Forest Tree Species in the Southeastern United States

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Abstract—Fire effects and behavior models that forecast postfire tree mortality include the First Order Fire Effects Model (FOFEM), BehavePlus, and the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS). Although these models are national in scope, the underlying equations driving the prediction of fire-related mortality were derived from data obtained from coniferous forests in the Western United States. For tree species common to upland forests of the Southeastern United States, quantitative models that predict mortality following prescribed fire are lacking. Consequently, these fire effects models utilize empirical models developed for western conifers to predict mortality of eastern deciduous and conifer species. Widespread application of models built with data outside the geographic range of application, let alone across functional groups, has the potential to introduce substantial error into model forecasts and misrepresent the ecological effects of prescribed burning. In this study, we (1) validated the equations used by nationally supported fire effects models to predict postfire mortality of upland tree species in the Southeastern United States using an extensive and geographically diverse independent dataset; and (2) developed new, species-specific postfire mortality models for some of the most common conifer and deciduous broadleaved tree species/species groups in upland forests of the Southeastern United States using easily obtained tree morphological and fire effects data. By developing postfire mortality for the suite of species that comprise upland forests, and incorporating those predictive models into available fire planning tools, managers will have improved ability to predict fire effects and assess the efficacy of burn efforts guided by restoration goals and objectives.

Keywords: Central Hardwood Region, oak mortality, pine mortality, prescribed burning, restoration.

INTRODUCTION

Fire effects models that forecast the effects of prescribed burning on various ecosystem attributes include the First Order Fire Effects Model v. 6.3 (FOFEM) (Reinhardt 2003), the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) (Rebain 2010), and BehavePlus v. 3.0 (Andrews and others 2005). Depending on species, these programs incorporate models that predict the probability of individual tree mortality as a function of stem size, bark thickness, and crown and/or bole damage. These computer simulation models are invaluable in that they allow resource managers to assess, prior to burning, the potential effects of

alternative prescribed burning prescriptions on a variety of stand-level variables including changes in structure via fire-related tree mortality. In 2014, almost 6.2 million acres of prescribed burning was conducted on forest land in the Southeastern United States (Melvin 2015). Accurate predictions of postfire mortality in the forests of this region are particularly important as restoration goals achieved via prescribed burning must consider effects on timber production and quality.

Although the aforementioned fire effects models are national in scope, the underlying equations driving the prediction of fire-related mortality were

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developed from data obtained from coniferous forests of the Western United States. Variables found to influence mortality of western conifer species such as crown damage (Ryan and Reinhardt 1988) have little predictive power for some eastern conifer species (Johansen and Wade 1987, Outcalt and Foltz 2004). Similarly, while metrics of crown damage are primary factors controlling mortality in western conifer species, they may not be the primary factor influencing mortality or topkill of deciduous broadleaved forest species (Catry and others 2010, Regelbrugge and Smith 1994), as the majority of prescribed burns in upland forests of the Southeastern United States are of low intensity (e.g., scorch/char heights <3 feet) (Hutchinson and others 2005) and rarely cause crown damage. Instead, fire effects related to cambial injury such as bole char severity rating (Kobziar and others 2006) or simple measurements of bole char height (Catry and others 2010) tend to be predictive of postfire mortality/topkill in deciduous broadleaved species.

With the exception of some species (e.g., *Pinus palustris*) (Varner and others 2007, Wang and others 2007)), quantitative models that predict mortality following prescribed fire specific to the >90 tree species found in upland southeastern forests, including both deciduous broadleaved and conifer species, are lacking. Following wildfire in the mountains of Virginia, Regelbrugge and Smith (1994) found diameter at breast height (DBH) and maximum height of bole char accurately predicted topkill of prominent upland *Quercus* species, *Acer rubrum*, *Nyssa sylvatica*, *Amelanchier arborea*, and *Carya glabra*. These equations, although some of the only fire-related mortality equations developed for deciduous broadleaved species, have not been validated for use following prescribed fire nor have they been tested for applicability outside of Virginia.

Because of the gap in knowledge regarding postfire tree mortality in forests of the Western versus Eastern United States, fire planning programs utilize empirical mortality models developed for western conifer species to predict and forecast mortality of eastern tree species. Widespread application of models built with data outside the geographic range of application and applied regardless of species has the potential to introduce substantial error into model forecasts, result in poor model performance,

and misrepresent the ecological effects of prescribed burning across multiple temporal and spatial scales. In this study, we (1) validated the equations used by nationally supported fire effects models to predict postfire mortality of upland tree species in the Southeastern United States using an extensive and geographically diverse independent dataset; (2) developed new, species-specific postfire mortality models for 17 conifer and deciduous broadleaved tree species/species groups common to upland forests of the Southeastern United States using easily obtained tree morphological and fire effects data; and (3) compared accuracy of the equations produced by Ryan and Amman (1994) and Regelbrugge and Smith (1994) with the newly developed equations. Availability of species-specific individual tree postfire mortality models, such as those developed in this study, is critical to developing burn prescriptions associated with restoration efforts and forecasting the effects of prescribed burning on short- and long-term species composition in upland forests of the Southeastern United States.

METHODS

Study Sites and Data Collection

A total of 244 0.25-acre (66 feet × 164 feet) plots were established in 94 prescribed burn units throughout 13 U.S. Department of the Interior, National Park Service (NPS) lands (fig. 1) using a standardized NPS vegetation monitoring protocol (USDI National Park Service 2003). Prescribed burns ranged in size from ~1 to 5,010 acres. Most burns were completed in the late dormant to early growing season (80 percent in March and April) between 1997 and 2012. Prior to burning, individual trees were tagged, and species, status (live/dead), and stem diameter at 4.5 feet above groundline (DBH; inches) were recorded. Overstory trees (stems >5.9 inches DBH) were inventoried in the entire 0.25-acre plot, and understory trees (stems ≥1.0 inch and ≤5.9 inches DBH) were inventoried in one-quarter of the larger plot. Direct fire effects for each tagged tree were measured within approximately 1 month post-burn and included scorch height (SCOR) and char height (CHAR). Scorch height (feet) was measured from the ground level to the highest point in the crown where foliar death was evident. Because the majority of the postfire inventories were conducted during leaf-off

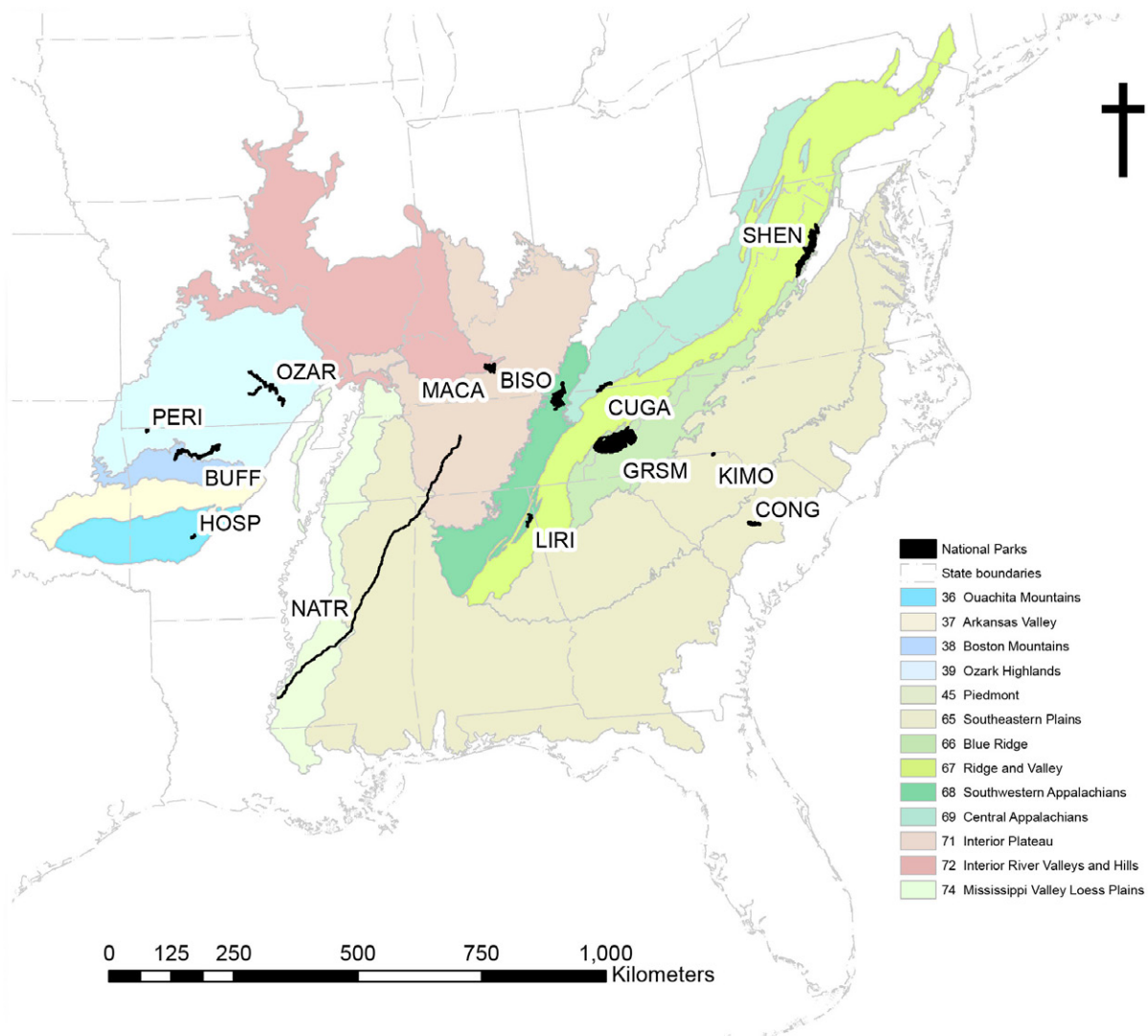


Figure 1—Map of study locations. BISO=Big South Fork National River and Recreation Area; GRSM=Great Smoky Mountains National Park; SHEN=Shenandoah National Park; CUGA=Cumberland Gap National Historic Park; MACA=Mammoth Cave National Park; KIMO=Kings Mountain National Military Park; OZAR=Ozark National Scenic Riverways; LIRI=Little River Canyon National Preserve; PERI=Pea Ridge National Military Park; CONG=Congaree National Park; NATR=Natchez Trace Parkway; HOSP=Hot Springs National Park; BUFF=Buffalo National River. Colors represent Level III Ecoregions (Omernik 1987).

(i.e., dormant season), SCOR measurements were obtained only for those deciduous broadleaved individuals possessing foliage. Char height (feet) was measured from the ground level to the highest point on the bole where char or scorch was evident regardless of slope (e.g., uphill, side hill, or downhill) position. Status (live/dead) of individual trees was recorded 2 years postfire with the exception of a subset of plots where individual trees were inventoried between 3 and 5 years postfire. In those instances, a tree was considered to be alive 2 years postfire if it was recorded as being alive

during the 3- to 5-year postfire inventory. This type of imputation was not performed for dead trees. Trees recorded as dead included those that were top-killed (i.e., main bole killed) and resprouting as a result of the fire.

Statistical Analyses

Validation—In FOFEM v. 6.3 (Lutes 2016), BehavePlus v. 3.0 (Andrews and others 2005), and, for all but eight species (*Quercus montana*, *Quercus alba*, *Quercus coccinea*, *Quercus rubra*, *Quercus velutina*, *Acer rubrum*, *Nyssa sylvatica*, and *Carya*,

in the Southern variant (SN) of FFE-FVS (Rebain 2010), the probability of postfire mortality for trees >1 inch DBH is modeled using the equation presented by Ryan and Amman (1994) (equation 1), which has the form:

$$P(m) = \frac{1}{1 + e^{(-1.941 + 6.316(1 - e^{-BT}) - 0.000535CS^2)}} \quad (1)$$

where:

$P(m)$ = probability of individual tree mortality (0-1)

BT = bark thickness (in)

CS = crown volume scorched (%).

Crown volume scorched is a function of crown length scorched which is estimated using measurements of scorch height or predicted flame length combined with measurements of total tree height and crown ratio (Van Wagner 1973). For eastern species other than *Pinus palustris*, BT is estimated as a linear relationship between a species-specific multiplier (Lutes 2016) and DBH .

In FFE-FVS (SN), the probability of mortality following fire of *Quercus alba*, *Quercus montana*, *Quercus coccinea*, *Quercus rubra*, *Quercus velutina*, *Nyssa sylvatica*, *Carya* species, and *Acer rubrum* is modeled using equation 2:

$$P(m) = \frac{1}{1 + e^{(\beta_0 + \beta_1 DBH + \beta_2 SBC)}} \quad (2)$$

where:

$P(m)$ = probability of individual tree mortality (0-1)

DBH = diameter at breast height (cm)

SBC = maximum point of bole blackening (m)

$\beta_0 - \beta_2$ are species-specific model coefficients presented by Regelbrugge and Smith (1994).

In FFE-FVS (SN), postfire mortality of *Quercus alba* is predicted using the equation specific to *Quercus montana*, and mortality of *Quercus coccinea*, *Quercus rubra*, and *Quercus velutina* is

modeled using the general *Erythrobalanus* equation (Regelbrugge and Smith 1994).

Using the independent NPS dataset described previously, we used the batch processing option in FOFEM v. 6.3 to calculate the probability of mortality as predicted by equation 1 for 28 species (table 1). Required input variables in the mortality module of FOFEM v. 6.3 include species, DBH , maximum scorch height, total tree height, and crown ratio. Maximum scorch height was entered as the maximum of $CHAR$ and $SCOR$. For deciduous broadleaved species, using the maximum of $SCOR$ or $CHAR$ may underestimate the effects of scorch, as $SCOR$ was only obtained when postfire inventories occurred during leaf-on. Because total tree height and crown ratio were not measured during data collection efforts, we used the SN, version 1860, to impute these variables (Keyser 2008). This adds a potential source of error in the validation effort. However, total height and canopy base height, two variables required to calculate crown ratio, are rarely collected in upland hardwood forests as part of normal ecological inventories. In addition, we used our independent dataset to predict the probability of mortality of *Quercus montana*, *Quercus alba*, *Quercus coccinea*, *Quercus rubra*, *Quercus velutina*, *Carya* species, *Acer rubrum*, and *Nyssa sylvatica* using equation 2 and model coefficients from Regelbrugge and Smith (1994).

Accuracy of equations 1 and 2 was assessed using classification tables (Hood and others 2007) and receiver operating characteristic (ROC) curve analysis (Saveland and Neuenschwander 1990). The ROC curve is a plot of the true positive rate (i.e., trees observed dead and predicted dead) versus the false positive rate across a range (0 to 1) of cutoff points (Hosmer and Lemeshow 2000). Models with ROC values ≥ 0.70 are considered to have an acceptable discrimination between live and dead trees, ROC values ≥ 0.80 have excellent discrimination, and ROC values ≥ 0.90 are considered to have outstanding discrimination (Hosmer and Lemeshow 2000). In the classification tables, trees whose predicted probability of mortality was above three specified cut-off values (0.50, 0.70, 0.90) were classified as dead while trees whose predicted probability of mortality was below the cutoff value were classified as live. Values in the classification tables reflect sensitivity

Table 1—Individual tree attributes of species used in the validation and/or development of postfire mortality models

Species	Number of live trees	Number of dead trees	DBH <i>inches</i> ^a	CHAR <i>feet</i> ^a	SCOR <i>feet</i> ^a
<i>Acer rubrum</i>	497	145	5.9 ± 3.6 (1.0, 27.6)	1.0 ± 1.6 (0.0, 11.5)	3.8 ± 11.0 (0.0, 98.4)
<i>Acer saccharum</i>	52	4	5.7 ± 4.0 (1.2, 25.1)	0.7 ± 1.3 (0.0, 5.4)	4.0 ± 8.1 (0.0, 29.5)
<i>Carya</i> species	524	49	8.8 ± 3.6 (1.0, 21.1)	1.1 ± 1.6 (0.0, 13.1)	2.1 ± 7.5 (0.0, 65.6)
<i>Cornus florida</i>	100	40	3.0 ± 2.1 (1.0, 9.4)	0.7 ± 0.9 (0.0, 6.6)	5.2 ± 9.1 (0.0, 36.1)
<i>Fraxinus</i> species	49	7	6.6 ± 4.5 (1.0, 20.3)	1.5 ± 2.6 (0.0, 11.5)	2.5 ± 6.8 (0.0, 29.5)
<i>Juniperus virginiana</i>	303	17	8.3 ± 3.2 (1.5, 18.0)	0.7 ± 2.1 (0.0, 16.4)	0.7 ± 3.6 (0.0, 32.8)
<i>Liquidambar styraciflua</i>	110	13	5.8 ± 4.0 (1.0, 17.3)	1.2 ± 2.8 (0.0, 22.3)	3.1 ± 9.1 (0.0, 65.6)
<i>Liriodendron tulipifera</i>	34	4	10.6 ± 6.9 (1.0, 34.3)	2.0 ± 2.6 (0.0, 9.8)	0.6 ± 4.0 (0.0, 24.6)
<i>Nyssa sylvatica</i>	234	105	5.2 ± 4.3 (1.0, 26.3)	2.4 ± 2.7 (0.0, 17.4)	2.5 ± 8.0 (0.0, 65.6)
<i>Ostrya virginiana</i>	31	4	3.0 ± 1.7 (1.0, 8.5)	0.6 ± 1.1 (0.0, 4.9)	1.3 ± 3.4 (0.0, 13.1)
<i>Oxydendrum arboreum</i>	308	55	5.3 ± 3.1 (1.0, 15.7)	1.9 ± 3.4 (0.0, 23.0)	7.5 ± 15.8 (0.0, 82.0)
<i>Pinus echinata</i>	524	40	11.1 ± 3.8 (1.1, 23.7)	7.4 ± 8.7 (0.0, 65.6)	10.7 ± 21.0 (0.0, 98.4)
<i>Pinus rigida</i>	220	43	11.3 ± 4.3 (1.3, 24.1)	4.0 ± 4.2 (0.0, 26.2)	2.7 ± 7.9 (0.0, 45.9)
<i>Pinus strobus</i>	174	43	8.7 ± 4.5 (1.0, 27.0)	1.6 ± 2.1 (0.0, 14.8)	5.4 ± 11.0 (0.0, 59.0)
<i>Pinus taeda</i>	63	13	11.2 ± 4.2 (2.0, 21.2)	6.7 ± 6.6 (0.0, 29.5)	9.8 ± 17.5 (0.0, 59.0)
<i>Pinus virginiana</i>	350	194	8.5 ± 3.5 (1.0, 21.0)	2.8 ± 3.5 (0.0, 23.0)	4.5 ± 11.6 (0.0, 65.6)
<i>Quercus alba</i>	593	43	11.3 ± 5.3 (1.0, 33.2)	1.6 ± 2.6 (0.0, 21.3)	2.5 ± 8.5 (0.0, 65.6)
<i>Quercus coccinea</i>	490	123	12.4 ± 5.0 (1.0, 32.4)	2.2 ± 2.6 (0.0, 32.8)	7.5 ± 19.5 (0.0, 98.4)
<i>Quercus falcata</i>	130	22	9.7 ± 5.7 (1.0, 30.2)	1.8 ± 3.9 (0.0, 36.1)	5.3 ± 11.7 (0.0, 49.2)
<i>Quercus marilandica</i>	119	71	7.5 ± 2.9 (1.1, 17.8)	2.0 ± 2.9 (0.0, 15.7)	7.0 ± 13.3 (0.0, 55.8)
<i>Quercus muehlenbergii</i>	48	2	7.8 ± 3.6 (1.6, 18.8)	0.4 ± 0.9 (0.0, 4.3)	0.0 ± 0.0 (0.0, 0.0)
<i>Quercus montana</i>	821	72	11.0 ± 5.1 (1.1, 38.2)	2.0 ± 3.1 (0.0, 23.0)	5.6 ± 15.2 (0.0, 82.0)
<i>Quercus rubra</i>	94	11	10.2 ± 4.0 (1.1, 20.9)	1.4 ± 1.7 (0.0, 6.6)	0.2 ± 1.8 (0.0, 13.1)
<i>Quercus stellata</i>	375	17	10.1 ± 4.1 (1.3, 28.5)	1.3 ± 2.2 (0.0, 19.7)	1.2 ± 6.5 (0.0, 65.6)
<i>Quercus velutina</i>	383	80	11.1 ± 5.6 (1.1, 30.0)	2.1 ± 3.0 (0.0, 39.4)	3.5 ± 11.0 (0.0, 82.0)
<i>Sassafras albidum</i>	31	43	2.2 ± 1.7 (1.0, 10.0)	2.0 ± 2.1 (0.0, 9.5)	1.8 ± 5.6 (0.0, 31.2)
<i>Tsuga canadensis</i>	35	10	6.9 ± 2.4 (1.7, 13.7)	1.1 ± 1.7 (0.0, 7.2)	10.4 ± 11.0 (0.0, 39.4)
<i>Ulmus</i> species	49	7	4.3 ± 2.6 (1.1, 10.0)	0.8 ± 1.5 (0.0, 5.6)	0.9 ± 4.3 (0.0, 29.5)

DBH=diameter at 4.5 feet above groundline; CHAR=maximum height of bole char; SCOR=maximum height of crown scorch.

^a Values represent the mean ± standard deviation (minimum, maximum).

(i.e., the proportion of dead trees correctly predicted to die within 2 years postfire) and associated false negative rate (FNR) and specificity (i.e., the proportion of live trees correctly predicted to live within 2 years postfire) and associated false positive rate (FPR). Sensitivity is referred as the true positive rate while specificity is often referred to as the true negative rate. The FPR, which is defined

as the proportion of live trees that were incorrectly classified as dead is calculated as 1-specificity. The FNR, which is the proportion of dead trees that were incorrectly classified as live, is calculated as 1-sensitivity. A high FPR indicates the model overpredicted mortality while a high FNR indicates the model underpredicted mortality.

Postfire mortality model development—We utilized the aforementioned dataset to develop new models that predict the probability of mortality following prescribed fire. Mortality data are categorical with a binary outcome (live or dead). Consequently, we used generalized linear mixed effects modeling to predict the probability of mortality 2 years post-fire using equation 3:

$$P(m) = \frac{1}{1 + e^{-(\beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_n X_n)}} \quad (3)$$

where:

$P(m)$ is the probability of mortality 2 years following prescribed burning

β_0 through β_n are the regression coefficients

X_1 through X_n are explanatory variables.

For both deciduous broadleaved and conifer species, tree-level explanatory variables tested were DBH and CHAR. These variables were utilized for a variety of reasons: (1) these variables are easily incorporated into the FOFEM and FFE-FVS modeling frameworks (e.g., CHAR is estimated to be 70 percent of modeled flame length (Rebain 2010); (2) DBH is a morphological variable easily obtained during pre- and postfire inventories; (3) relationships between DBH and bark thickness, a variable that may better predict postfire mortality, are unavailable for many upland tree species; and (4) CHAR is a fire effects variable readily obtained during postfire inventories. Models were fitted using maximum likelihood methods and the adaptive Gaussian Quadrature method (METHOD=QUAD) in PROC GLIMMIX (SAS Institute 2015). Plot was included as random effects to account for the fact that trees were nested within a plot. Species-specific models were developed when the number of events (i.e., dead trees) was $\geq 5p$, where p is the number of explanatory variables (Vittinghoff and McCulloch 2007). Adequate sample size, therefore, equated to a minimum of 10 dead trees (table 1). Model goodness-of-fit was assessed using ROC curve analysis (Saveland and Neuenschwander 1990) using the conditional (i.e., incorporating random effects) predicted values. As a caveat, postfire mortality dynamics for *Pinus virginiana*

in this study could have been influenced by a *Dendroctonus frontalis* outbreak that occurred within the Great Smoky Mountains National Park in eastern Tennessee during the late 1990s.

RESULTS

Validation

At the 0.50 cutoff value, which is most commonly used to signal mortality (Lutes 2016), total accuracy of equation 1 varied from 11 percent for *Ostrya virginiana* and *Acer saccharum* to 71 percent for *Quercus montana* and *Pinus taeda* (table 2). At the 0.50 cutoff, model sensitivity was varied between 21 and 100 percent. Although sensitivity values for the majority of species were above 50 percent, specificity, which provides information on how well equation 1 predicted survival, was comparatively low. The false positive rate (i.e., 1-specificity) of equation 1 at the 0.50 cutoff level was relatively high, indicating equation 1 vastly overpredicted mortality for the species examined. Total accuracy of equation 1 increased as the cutoff value increased for all species examined other than *Sassafras albidum*. The increase in total accuracy achieved by using a higher cutoff value corresponded to decrease in model sensitivity and an increase in model specificity. Of the 28 species validated, only 10 were found to have acceptable levels of discrimination (i.e., ROC values ≥ 0.70) (table 2).

The ROC values associated with equation 2, which was developed using data specific to the species modeled, indicated acceptable to excellent discrimination (table 3). Total accuracy at the 0.50, 0.70, and 0.90 cutoff values exceeded 70 percent for all eight species examined. Across the cutoff values, survival was more accurately predicted than mortality. At the standard 0.50 cutoff value, sensitivity was greatest for *Quercus velutina* (69 percent) while specificity was greatest for *Nyssa sylvatica* (99 percent). The ROC values indicated that relative to equation 1, equation 2 performed similarly for *Acer rubrum* (ROC 0.807 versus 0.817), but outperformed equation 1 for *Carya* species, *Nyssa sylvatica*, *Quercus alba*, *Quercus coccinea*, *Quercus montana*, *Quercus rubra*, and *Quercus velutina*. At the standard 0.50 cutoff value, the FPR, or overprediction of mortality, of species

Table 2—Classification table representing the validation of the Ryan and Amman (1994) postfire mortality model (equation 1) as implemented in the FOFEM v. 6.3 (Lutes 2016) at 0.50, 0.70, and 0.90 cutoff values

Species	ROC	Total accuracy	Sensitivity (FNR)	Specificity (FPR)	Total accuracy	Sensitivity (FNR)	Specificity (FPR)	Total accuracy	Sensitivity (FNR)	Specificity (FNR)
		0.50 cutoff value			0.70 cutoff value			0.90 cutoff value		
<i>Acer rubrum</i> ^a	0.817	26	100 (0)	5 (95)	56	92 (8)	46 (54)	76	6 (94)	97 (3)
<i>Acer saccharum</i>	0.800	11	100 (0)	4 (96)	52	100 (0)	48 (52)	93	14 (86)	100 (0)
<i>Carya species</i> ^a	0.726	49	67 (37)	47 (53)	91	51 (49)	95 (5)	92	14 (86)	99 (1)
<i>Cornus florida</i>	0.568	29	100 (0)	1 (99)	42	88 (12)	24 (76)	63	13 (87)	83 (17)
<i>Fraxinus species</i>	0.832	38	100 (0)	29 (71)	73	100 (0)	69 (31)	89	14 (86)	100 (0)
<i>Juniperus virginiana</i>	0.851	34	88 (12)	31 (69)	87	76 (24)	87 (13)	94	6 (94)	99 (1)
<i>Liquidambar styraciflua</i>	0.663	24	92 (8)	15 (85)	57	77 (23)	55 (45)	87	8 (92)	96 (4)
<i>Liriodendron tulipifera</i>	0.901	61	100 (0)	56 (44)	87	75 (25)	88 (12)	87	0 (100)	97 (3)
<i>Nyssa sylvatica</i> ^a	0.796	42	100 (0)	17 (73)	68	92 (8)	56 (44)	67	2 (98)	97 (3)
<i>Ostrya virginiana</i> ^a	0.577	11	100 (0)	0 (100)	17	75 (25)	10 (90)	89	0 (100)	100 (0)
<i>Oxydendrum arboreum</i>	0.651	17	98 (2)	3 (97)	53	67 (33)	50 (50)	80	24 (76)	90 (10)
<i>Pinus echinata</i>	0.613	52	65 (35)	51 (49)	85	30 (60)	90 (10)	87	30 (70)	92 (8)
<i>Pinus rigida</i>	0.528	63	21 (79)	72 (28)	81	2 (98)	96 (4)	83	0 (100)	100 (0)
<i>Pinus strobus</i>	0.721	56	72 (28)	52 (48)	78	42 (59)	87 (13)	85	42 (58)	96 (4)
<i>Pinus taeda</i>	0.569	71	38 (62)	78 (22)	78	0 (100)	94 (6)	79	0 (100)	95 (5)
<i>Pinus virginiana</i>	0.630	48	92 (8)	24 (76)	62	17 (83)	87 (13)	64	6 (94)	96 (4)
<i>Quercus alba</i> ^a	0.695	60	81 (19)	58 (42)	90	16 (84)	96 (4)	92	5 (95)	98 (2)
<i>Quercus coccinea</i> ^a	0.652	68	58 (42)	71 (29)	76	25 (75)	89 (11)	77	15 (85)	93 (7)
<i>Quercus falcata</i>	0.530	52	55 (45)	52 (48)	74	36 (64)	81 (19)	85	14 (96)	97 (3)
<i>Quercus marilandica</i>	0.731	51	99 (1)	23 (77)	66	44 (56)	80 (20)	66	28 (72)	89 (11)
<i>Quercus muehlenbergii</i>	0.609	32	50 (50)	31 (69)	78	0 (100)	81 (19)	96	0 (100)	100 (100)
<i>Quercus montana</i> ^a	0.666	71	54 (46)	73 (27)	86	28 (72)	91 (9)	88	3 (97)	95 (5)
<i>Quercus rubra</i> ^a	0.654	63	64 (36)	63 (37)	91	36 (64)	98 (2)	90	0 (100)	100 (100)
<i>Quercus stellata</i>	0.574	56	53 (47)	56 (44)	93	12 (88)	96 (4)	94	0 (100)	99 (1)
<i>Quercus velutina</i> ^a	0.724	68	60 (40)	70 (30)	82	35 (65)	92 (8)	79	10 (90)	94 (6)
<i>Sassafras albidum</i>	0.619	58	100 (0)	0 (100)	62	98 (2)	13 (87)	49	14 (86)	97 (3)
<i>Tsuga canadensis</i>	0.657	31	100 (0)	11 (89)	69	40 (60)	77 (23)	80	30 (70)	94 (6)
<i>Ulmus species</i>	0.545	14	100 (0)	2 (98)	32	86 (14)	24 (76)	88	0 (100)	100 (0)

ROC=area under the Receiver Operating Characteristics curve; Total accuracy=the percentage of trees correctly classified as either live or dead; Sensitivity (FNR)=the percentage of trees observed dead that were correctly predicted to die within 2 years postfire (false negative rate (1-sensitivity)); Specificity (FPR)=the percentage of trees observed live that were correctly predicted to survive 2 years postfire [false positive rate (1-specificity)].

^a Indicates the probability of mortality calculated in FFE-FVS (SN) is achieved via equation 2.

modeled using equation 2 was substantially lower than equation 1.

Postfire Mortality Models

For *Liquidambar styraciflua*, *Pinus rigida*, *Pinus taeda*, *Quercus rubra*, and *Tsuga canadensis*, neither DBH nor CHAR were significant predictors of postfire mortality. For all species other than *Quercus falcata* and *Quercus stellata*, we observed a significant and negative relationship between the probability of mortality and DBH (table 4). This negative relationship implies the probability of mortality within 2 years following prescribed fire decreases as DBH increases. For *Juniperus virginiana* and *Pinus echinata*, CHAR had no significant relationship with the probability of mortality. For all other species, the probability of mortality was significantly and positively related to CHAR (table 4). With the exception of *Quercus alba*, which had an acceptable level of discrimination (ROC=0.795), ROC values indicated the newly developed postfire models had excellent (ROC >0.8) to outstanding (ROC >0.9) discrimination.

DISCUSSION

The goals of this study were to (1) validate existing postfire mortality models used by nationally supported fire behavior and fire effects models; (2) develop new species-specific postfire mortality models for prominent upland forest tree species in the Southeastern United States using individual tree and fire effects data commonly collected during postfire inventory procedures; and (3) compare accuracy of the equations produced by Ryan and Amman (1994) and Regelbrugge and Smith (1994) with the newly developed equations.

Validation

The use of postfire mortality models developed for western conifer species (equation 1) inadequately predicted postfire mortality of the eastern tree species examined in this study. Equation 1 overpredicted mortality overall and by as much as 100 percent (using the standard 0.50 cutoff) for some species (e.g., *Ostrya virginiana* and *Sassafras albidum*). Not surprisingly, the postfire mortality equations developed specifically for a subset of the tree species examined in this study

(equation 2) more accurately predicted mortality than equation 1. For example, at the standard 0.50 cutoff value, equation 1 correctly predicted mortality (i.e., sensitivity) of 54 percent of the *Quercus montana* trees in our dataset compared to 64 percent using equation 2. For the majority of species examined, equation 1 predicted mortality more accurately than survival (i.e., specificity), while equation 2 predicted survival more accurately than mortality. Our results support the notion that widespread application of models built with data outside the geographic range of application, let alone across functional groups (i.e., conifer versus deciduous broadleaved species), results in poor model performance (Wang and others 2007). Model performance errors of the magnitudes observed in this study have the potential to misrepresent the ecological effects of prescribed burning on species composition and stand density—two attributes actively managed through prescribed burning programs across upland forests of the Southeastern United States.

Postfire Mortality Models

Stem size (i.e., DBH) and severity of fire effects (i.e., CHAR), singularly or in concert, significantly predicted the probability of individual tree mortality 2 years following prescribed fire for all species other than *Liquidambar styraciflua*, *Pinus rigida*, *Pinus taeda*, *Quercus rubra*, and *Tsuga canadensis*, (i.e., neither DBH or CHAR were significant). For 16 of the 17 species examined, we observed a significant and inverse relationship between the probability of mortality following prescribed burning and DBH. Stem size has been proven to be a significant predictor of postfire mortality of numerous eastern and western tree species (Beverly and Martell 2003, Hood and Bentz 2007, Hood and others 2010). Although significant, the strength of the relationship between the probability of mortality and DBH varied considerably across species, and was likely a reflectance of species-specific relationships between stem size and bark thickness (Harmon 1984). Individuals with larger DBH, and consequently greater bark thickness, experience lower maximum cambial temperatures and an increased time to reach peak temperature, providing greater protection from cambial injury (Hengst and Dawson 1994) and subsequent mortality.

Crown damage is commonly used to predict mortality of western conifer species (Hood and others 2010, Keyser and others 2006). Because the vast majority of prescribed burns in the southeastern upland forests are conducted during the dormant season, estimates of crown/foilage damage are rarely possible to obtain for deciduous broadleaved species. In addition, because maximum scorch height associated with prescribed burns in upland forests of the Southeastern United States is low (<3 feet) (Arthur and others 2015), crown damage is rarely observed following prescribed burning for either conifer or deciduous broadleaved species. Consequently, we fitted postfire mortality models using CHAR in lieu of crown damage or maximum height of crown scorch (SCOR) values. The probability of mortality for all the deciduous broadleaved species and all but two conifer species (*Juniperus virginiana* and *Pinus echinata*) modeled was positively associated with CHAR. Maximum height of bole char, which is an indirect measurement of cambial damage (Wyant and others 1986) and related to flame length (Rebain 2010), is positively correlated with tree mortality (Arthur and others 2015) and has been successfully used to predict mortality of conifer and deciduous broadleaved species, including *Pinus ponderosa* (Regelbrugge and Conard 1993), *Populus tremuloides* (Hély and others 2003), *Quercus kelloggii* (Cocking and others 2012), and various Appalachian hardwood species (Regelbrugge and Smith 1994). When crown damage and bole char height are used in combination to predict postfire mortality, crown damage is usually the stronger of the two predictors (Rigolot 2004). However, in systems where crown damage is negligible or cannot be accurately measured (e.g., as is the case for dormant deciduous broadleaved species), results from previous studies (Cocking and others 2012, Regelbrugge and Smith 1994) coupled with results presented here suggest bole char height adequately predicts postfire tree mortality.

CONCLUSION

The equations developed in this study outperformed the general Ryan and Amman (1994) equations for all species examined, suggesting models used by nationally supported fire effects and behavior models to predict postfire tree mortality should be developed using data specific to a given species or

species group obtained from the geographic area of interest. Furthermore, models that use fire effects data to predict mortality should include fire effects data relevant to a given species or species group. Models utilized by fire effects programs, including FOFEM and FFE-FVS, as well as the models developed in this study are rudimentary and utilize the most basic of fire effects (i.e., DBH and crown damage as predicted by flame length or scorch height) to predict postfire mortality. The probability a tree dies following fire is confounded by many factors other than scorch/char height and tree size, including prefire growth rates (van Mantgem and others 2003), fine-root mortality (Swezy and Agee 1991), season of burn (Harrington 1993), postfire insect activity (Menges and Deyrup 2001), tissue necrosis (Bova and Dickinson 2005), and abiotic conditions at the time of burn [e.g., duff moisture (Varner and others 2007)].

The newly developed postfire mortality models presented here used easily obtained tree and fire effects data, and should be considered a step towards the development of short- and long-term fire-related mortality models as related to prescribed burning for eastern tree species. The new equations presented in this study were developed using data from low-severity prescribed fire conducted primarily during the dormant season. Therefore, it is not known how well these equations will perform for trees damaged by higher severity prescribed or wildfire, or following growing season burn events. However, the high levels of model discrimination displayed by equation 2, which was developed using data following wildfire, suggests separate models for predicting postfire mortality following prescribed versus wildfire may not be necessary, but this should be a subject of future investigation. The models we developed forecast mortality 2 years postfire. Evidence from western (Thies and others 2006) and eastern (Yaussy and Waldrop 2008) systems document a delay in mortality that can extend through the fourth year following prescribed fire. Lack of more detailed fire effects and fire behavior data in combination with longer tree mortality data limited our ability to explore more complex relationships among delayed tree mortality and the myriad of fire effects known to influence postfire mortality. To improve upon or expand the use of the models developed in this study, we recommend collecting additional data

Table 3—Classification table representing the validation of the Regelbrugge and Smith (1994) postfire mortality model (equation 2) as implemented in the FFE-FVS (SN) (Rebain 2010) at 0.50, 0.70, and 0.90 cutoff values

Species	ROC	Total accuracy	Sensitivity (FNR)	Specificity (FPR)	Total accuracy	Sensitivity (FNR)	Specificity (FPR)	Total accuracy	Sensitivity (FNR)	Specificity (FPR)
		<i>0.50 cutoff value</i>			<i>0.70 cutoff value</i>			<i>0.90 cutoff value</i>		
<i>Acer rubrum</i>	0.807	78	48 (52)	42 (58)	76	24 (76)	91 (9)	75	11 (89)	94 (6)
<i>Carya species</i>	0.805	92	53 (47)	95 (5)	91	29 (71)	97 (3)	92	14 (86)	99 (1)
<i>Nyssa sylvatica</i>	0.886	79	34 (66)	99 (1)	75	19 (81)	100 (0)	71	8 (92)	100 (0)
<i>Quercus alba</i>	0.798	92	51 (49)	95 (5)	94	47 (53)	97 (3)	94	21 (79)	99 (1)
<i>Quercus coccinea</i>	0.734	77	54 (46)	83 (17)	80	41 (59)	90 (10)	81	24 (76)	95 (5)
<i>Quercus montana</i>	0.866	92	64 (36)	94 (6)	93	45 (55)	97 (3)	93	23 (77)	99 (1)
<i>Quercus rubra</i>	0.743	77	45 (55)	81 (19)	82	18 (82)	89 (11)	90	18 (82)	98 (2)
<i>Quercus velutina</i>	0.796	75	69 (31)	77 (23)	81	58 (42)	86 (14)	83	35 (65)	92

ROC=area under the Receiver Operating Characteristics curve; Total accuracy=the percentage of trees correctly classified as either live or dead; Sensitivity (FNR)=the percentage of trees observed dead that were correctly predicted to die within 2 years postfire (false negative rate (1-sensitivity)); Specificity (FPR)=the percentage of trees observed live that were correctly predicted to survive 2 years postfire [false positive rate (1-specificity)].

Table 4—Parameter estimates (standard error) associated with species-specific models for predicting 2-year mortality following a single prescribed burn

Species	β_0	β_1	β_2	ROC
<i>Acer rubrum</i>	1.5665 (0.5419)	-0.7910 (0.1082)	0.3600 (0.1180)	0.900
<i>Carya species</i>	-1.7070 (0.7405)	-0.3077 (0.0740)	0.3524 (0.1301)	0.964
<i>Cornus florida</i>	-0.9436 (0.4995)	-0.3741 (0.1459)	1.2042 (0.3519)	0.831
<i>Juniperus virginiana</i>	4.4307 (3.7436)	-1.9250 (0.6086)	NS	0.994
<i>Nyssa sylvatica</i>	2.0139 (0.7099)	-1.4082 (0.2358)	0.4579 (0.1311)	0.974
<i>Oxydendrum arboreum</i>	0.6011 (0.8520)	-0.9661 (0.2258)	0.5513 (0.1200)	0.957
<i>Pinus echinata</i>	-2.2510 (1.0604)	-0.2774 (0.0992)	NS	0.975
<i>Pinus strobus</i>	-3.0327 (1.6098)	-0.2673 (0.1327)	0.7908 (0.2683)	0.983
<i>Pinus virginiana</i>	0.8526 (0.6354)	-0.3855 (0.0598)	0.2359 (0.0556)	0.929
<i>Quercus alba</i>	-1.7759 (0.4147)	-0.1564 (0.0443)	0.2621 (0.0432)	0.795
<i>Quercus coccinea</i>	-1.6085 (0.4826)	-0.0945 (0.0315)	0.2262 (0.0638)	0.917
<i>Quercus falcata</i>	-3.3149 (0.9240)	NS	0.2643 (0.1305)	0.960
<i>Quercus marilandica</i>	0.2854 (0.8636)	-0.2497 (0.1094)	0.4615 (0.1240)	0.906
<i>Quercus montana</i>	-1.6320 (0.5843)	-0.3651 (0.0609)	0.4112 (0.0702)	0.951
<i>Quercus stellata</i>	-4.2562 (0.5525)	NS	0.3635 (0.0895)	0.915
<i>Quercus velutina</i>	0.1500 (0.5486)	-0.3311 (0.0670)	0.3848 (0.0859)	0.916
<i>Sassafras albidum</i>	1.3997 (0.6211)	-2.1525 (0.5236)	2.4615 (0.5756)	0.935

Model coefficients: β_0 =intercept; β_1 =diameter at breast height (inches) (DBH); β_2 =maximum height of stem char (feet) (CHAR); ROC=area under the Receiver Operator Characteristics curve; NS=not significant.

related to (1) longer term postfire mortality and severity of fire effects from the range of forest types and resultant species compositions, (2) edaphoclimatic conditions, and (3) prescribed burning goals and objectives inherent to upland forests of the Southeastern United States, including season of burn.

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Estimating Canopy Bulk Density and Canopy Base Height for Conifer Stands in the Interior Western United States Using the Forest Vegetation Simulator Fire and Fuels Extension

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Author's note: This is a summary of work that is completely described in Ex and others (2016).

The Forest Vegetation Simulator Fire and Fuels Extension (FFE-FVS) is often used to estimate canopy bulk density (CBD) and canopy base height (CBH), which are key indicators of crown fire hazard for conifer stands in the Western United States. Estimated CBD from FFE-FVS is calculated as the maximum 4 m running mean bulk density of predefined 0.3 m thick canopy layers (Sando and Wick 1972). Canopy base height is estimated in a similar fashion as the lowest height at which the running mean bulk density of canopy layers exceeds a predefined threshold of 0.011 kg m⁻³ (Scott and Reinhardt 2001). Because estimates of CBD and CBH from FFE-FVS are derived from estimates of the bulk density of canopy layers, their values depend both on the biomass of canopy fuel and on the manner in which fuel is distributed vertically within the crowns of trees that make up the canopy (Keyser and Smith 2010).

In this work, we evaluated the impact of using alternative crown fuel distributions and crown fuel biomass allometries on CBD and CBH estimation using FFE-FVS. We used the southwestern ponderosa pine sub-model of version 1108 of the Central Rockies (CR) Variant of FVS (Keyser and Dixon 2008) for our analysis. Our approach was to estimate CBD and CBH for mostly pure, even-aged stands of seven conifer species by modifying FFE-FVS to use non-uniform instead of uniform crown fuel distributions, which allowed us to

determine whether distribution effects on CBD and CBH estimates were species-specific or general. For two species, we also compared estimates derived using local versus non-local crown fuel biomass allometries to ascertain whether there was a consistent bias in CBD and CBH estimates associated with application of allometries outside their geographic area of origin.

We used crown biomass data from 319 trees in 59 mostly pure, even-aged conifer stands to evaluate the effects of using non-uniform crown fuel biomass distributions on CBD and CBH estimates. Stands were selected to represent broad ranges of average tree size and stand density for each species. Our data come from stands with quadratic mean diameters ranging from 3.3–43.7 cm and densities ranging from 136–25,542 trees ha⁻¹. Coordinates and physical characteristics of most of the stands, which were located throughout the interior Western United States, are reported in Ex and others (2015). Field methods and the remaining stands are described in detail in Ex and others (2015), Long and Smith (1988) and Long and Smith (1989). Data from a subset of 12 of the 59 stands (30 trees) were used to evaluate whether there was consistent bias in CBD and CBH estimates from FFE-FVS that was associated with geographic area. The allometries in FFE-FVS were developed for stands in Montana and northern Idaho (Brown 1978). We developed corresponding allometries using data from ponderosa pine and Douglas-fir stands located in Colorado, New Mexico, Utah, and southern Idaho. Using the non-uniform fuel

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distributions and local biomass allometries, we modified the CBD and CBH calculation procedure in FFE-FVS in three ways: (1) we incorporated non-uniform distributions, but retained crown biomass allometries from Brown (1978); (2) we retained the uniform distributions from the production version of FFE-FVS but incorporated local biomass allometries, and; (3) we incorporated both non-uniform distributions and local biomass allometries. For each cover type, we obtained estimates of CBD and CBH using our modifications and compared them to estimates from the production version of FFE-FVS.

The data showed estimates of CBD generated using non-uniform crown fuel biomass distributions were consistently 13–27 percent larger than estimates from the production version of FFE-FVS. The difference was statistically significant for all cover types except pinyon-juniper (table 1). Unlike CBD, estimates of CBH did not always increase. Average differences between estimates of CBH from the production version of FFE-FVS and from versions that used non-uniform crown fuel distributions ranged from -11 percent to +23 percent and were in most cases non-significant (table 1).

Although estimates of CBD and CBH generated using local crown fuel biomass allometries were sometimes substantially different than estimates from the production version of FFE-FVS, there was no statistical difference between estimates from the different methodologies (table 1). This was because in some stands estimates of canopy fuel load from local allometries were larger than estimates from non-local allometries, causing estimates of CBD to increase and potentially causing estimates of CBH to decrease, while in other stands the opposite was true (fig. 1). This suggests allometric relationships vary widely among stands in the southern Rockies.

The major implication of the consistent increase in estimated CBD we observed is a subsequent decrease in estimates of the critical spread rate required to sustain the spread of fire from tree to tree through canopies from fire behavior models (Scott and Reinhardt 2001). An exploratory analysis using our data suggested this decrease was generally on the order of 3 m min⁻¹, but it varied considerably among stands. Non-uniform distributions unquestionably offer more realistic representations of crown fuel distribution than uniform distributions. However, it is not clear that incorporating them in FFE-FVS will improve

Table 1—Average difference in estimated CBD and CBH from modified and production versions of FFE-FVS (average percentage change follows each value in parentheses). Values were calculated as modified – production.

Cover type ^a	FFE-FVS modification	CBD Δ (kg m ⁻³)	CBH Δ (m)
Ponderosa pine	Local allometries	0.023 (16)	-0.46 (-17)
Douglas-fir		0.034 (21)	-0.25 (-19)
Subalpine fir	Non-uniform distributions	0.074 (23)*	0.24 (22)
Ponderosa pine		0.028 (27)*	0.64 (17)*
Pinyon-juniper		0.019 (13)	-0.10 (-11)
Lodgepole pine		0.018 (13)*	0.19 (9)*
Engelmann spruce		0.055 (17)*	0.15 (23)
Douglas-fir		0.038 (24)*	0.41 (18)
Ponderosa pine	Local allo. & Non-uni. dist.	0.053 (48)*	-0.10 (-9)
Douglas-fir		0.066 (44)*	-0.05 (-7)

^aAfter Eyre and others (1980), excepting the Engelmann spruce-subalpine fir type which has been split into its constituent species here.

Significant differences from zero at $\alpha = 0.05$ are denoted by *.

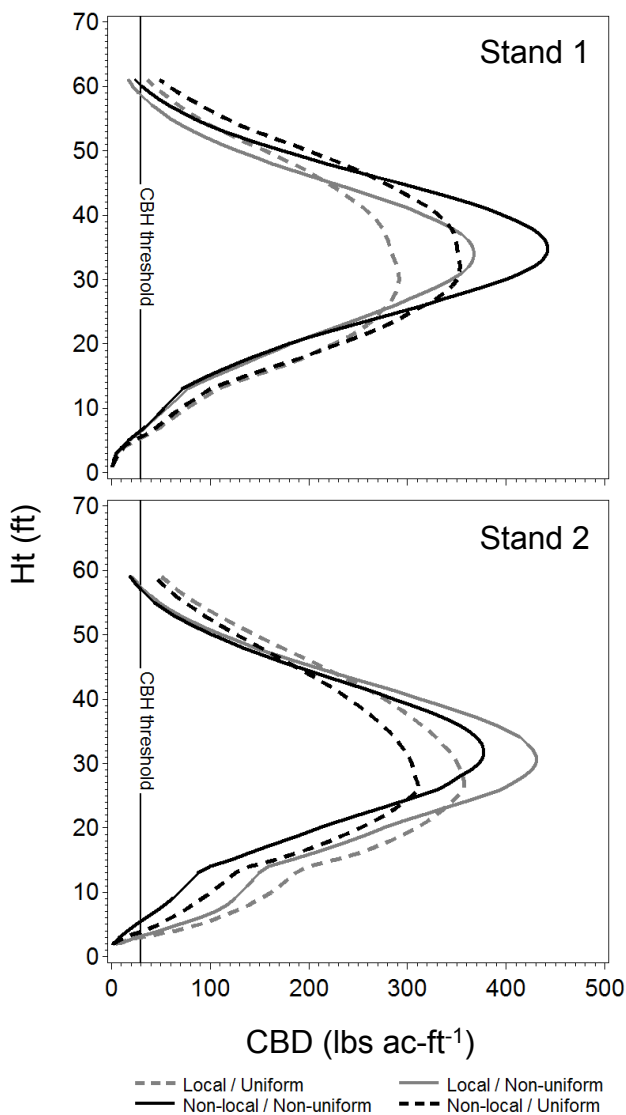


Figure 1—Canopy fuel profiles for two Douglas-fir stands created using modified CBD and CBH calculations that incorporated local biomass allometries and uniform crown fuel distributions, local allometries and non-uniform distributions, non-local allometries and uniform distributions, and non-local allometries and non-uniform distributions. Local allometries decreased the estimate of crown fuel biomass for Stand 1 and increased the estimate for Stand 2.

predictions of potential fire behavior unless the fire behavior and spread models in FFE-FVS are also re-parameterized for compatibility with the improved canopy fuel characterization methodology (Cruz and Alexander 2010). Percent changes in CBH from incorporating non-uniform distributions can be of a similar order of magnitude as changes in

CBD and occasionally much larger, but the direction and amount of change are difficult to predict for a given stand. This highlights the need to re-evaluate the method used to delineate CBH in FFE-FVS, as it is clearly sensitive to assumptions regarding the distribution of fuel within tree crowns.

Adopting local biomass allometries in FFE-FVS could potentially change estimates of CBD and CBH as much as adopting non-uniform crown fuel distribution assumptions. However, good estimates of CBD and CBH for southern Rocky Mountain stands require the use of allometric models that are capable of accounting for stand to stand variation in the relationship between d.b.h. and crown fuel biomass. This will likely require incorporating tree height or live crown ratio as predictor variables in allometric models, and argues for routine measurement of both tree and live crown base heights during inventories to permit use of allometries that incorporate this information (Tinkham and others, in press).

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Sensitivity of Crown Fire Modeling to Inventory Parameter Dubbing in FVS

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Abstract—Most forest restoration treatments in the Western United States seek to reduce crown fire potential, which is commonly evaluated in the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) to understand the effects of alternative treatments. Typically, tree characteristics needed to estimate stand attributes like canopy base height and canopy bulk density are inventoried on a subsample of trees and then are estimated (dubbed) through allometry for trees missing records. This study evaluates the effect of subsample tree intensity with height or height/crown ratio records on FFE-FVS estimate accuracy of crown fire behavior metrics in the Central Rockies (CR) variant. Results show FFE-FVS provides consistent estimates of canopy bulk density across the range of subsampled tree intensities, but found a systematic overestimation bias for FVS predicted tree crown ratios and subsequently stand canopy base height that was overcome by the inclusion of crown ratio observations. Therefore, inventories seeking to model crown fire potential in FFE-FVS should consider increasing the inclusion of crown ratio observations for sample trees.

INTRODUCTION

There is growing concern in the management of dry-mixed conifer forests of the Western United States about restoration of heterogeneous spatial patterns and processes, while still facilitating the reduction of potential crown fire behavior (Underhill and others 2014). This reduction in fire behavior has grown in emphasis over the last couple of decades following a series of widespread, high severity fires throughout the region (Lannom and others 2014). Increasingly, the demand has shifted towards fuels reduction treatments that are designed not only to meet these fire behavior requirements, but also help restore historical spatial forest structure and function (Tinkham and others 2016a). Research utilizing both empirical observations and modeling have been conducted to evaluate how forest structures impact crown fire behavior (Fulé and others 2012, Hudak and others 2009, Stephens and others 2009). Previous studies have focused on quantifying two transition points that need to be characterized to capture crown fire behavior, including the movement of fire from the surface to the crown and then the horizontal propagation of fire from crown to crown.

The vertical transition of a surface fire into a crown is controlled by surface fuel loading, canopy base height (CBH), wind speed, and slope, this phenomenon is commonly referred to as torching or passive crown fire (Scott and Reinhardt 2001). In passive crown fires, individual or small groups of crowns will torch but will not be sustained; while in active crown fires the entire crown fuel strata is engaged, but where the sustainability of the crown fire is reliant on the heat released from the combustion of the surface fuels (Scott and Reinhardt 2001). Within fire management this concept is captured through the characterization of a Torching Index which can be defined as the 10 m open wind speed necessary for the onset of passive crown fire behavior. The second transition is the propagation of fire from crown to crown across a stand which is controlled by crown bulk density (CBD) and wind speed, and is referred to as crowning. This concept is characterized in fire management through the Crowning Index which is defined as the 10 m open wind speed necessary for the onset of active crown fire behavior (Scott and Reinhardt 2001).

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Research has characterized these phenomena and described how management can manipulate three physical features of the forest environment to influence crown fire behavior, including surface fuel loading, CBH, and CBD (Hudak and others 2009, Scott and Reinhardt 2001). While it is understood how manipulating these elements of a stand changes fire behavior, less attention has been paid to what it takes to accurately characterize these features for modeling treatment alternatives. When it comes to fuels sampling, there is a growing body of literature describing new techniques for improving sampling accuracy (Keane 2013, Tinkham and others 2016b) and highlighting the need to consider surface fuel spatial variability (Keane and others 2012, Smith and others 2017, Vakili and others 2016). Currently, most modeling systems available to management are unable to consider the effects of fuel spatial variability on the physical relationship controlling the transition of a surface fire to passive crown fire activity (Hudak and others 2009). When it comes to the characterization of CBD and CBH within a growth and yield model, there is limited literature describing how the sampling protocols or the variability in forest structure affects accuracy of predicted metrics and what this means for crown fire behavior modeling (Cruz and others 2004, Leites and others 2009).

One of the most widely used models for evaluating treatment alternatives in these systems is the U.S. Department of Agriculture Forest Service's Forest Vegetation Simulator (FVS), which is a distance-independent growth and yield model operating at the individual tree level (Crookston and Dixon 2005, Dixon 2002). The FVS system is capable of making stand level estimates of forest structure from a minimal set of required parameters including geographic model variant, sample design, and tree diameter at breast height (1.37 m above groundline; DBH). From these required parameters, a "dubbing" process based on allometric equations developed for each geographic model variant is used to fill in missing information such as individual tree heights and crown ratios (Shaw and others 2006). Dubbing is the processes of predicting missing values from ancillary data and the FVS system has built in dubbing equations for multiple parameters including height, crown length, and crown ratio (Bragg 2008, Keyser and Dixon 2008). Within the FVS-Central Rockies variant (CR), missing height values are

estimated following equation 1, where coefficients B_1 and B_2 are species-specific and user provided height data ($n \geq 3$) are used to calculate a site specific B_1 coefficient (Keyser and Dixon 2008). Other tree level metrics like crown length (equation 2) and crown ratio (equation 3) use equations with species-specific coefficients to fill missing records, but these equations are not adaptive to the inventory dataset. This dubbing process makes it possible to estimate characteristics like CBD and CBH at the stand level.

$$Height (ft) = 4.5 + \exp\left(B_1 + \frac{B_2}{DBH (inches)+1}\right) \quad (1)$$

$$Crown Length (ft) = B_0 + (B_1 \times Height) + (B_2 \times DBH) + (B_3 \times Basal \text{ area per acre}) \quad (2)$$

$$Crown Ratio = \frac{Crown Length}{Height} \quad (3)$$

The forest types within CR occupy a large and varied geographic extent, where forest types like the ponderosa pine woodland ecosystem occupy sites that vary greatly in productivity, climate, and management (Keyser and Dixon 2008) (fig. 1). Development of the CR variant was based on the GENeralized Growth and Yield Model (GENGYM) stand projection system that utilized a network of 359 temporary and permanent growth monitoring plots collected on National Forest land in the 1970s and 1980s for all ecosystems (Edminster and others 1991). Of these plots, only 30 were in ponderosa pine (*Pinus ponderosa*) dominated stands north of Arizona and New Mexico, on the San Juan National Forest, and studies focusing on validating the performance of the model are limited.

A consideration for most forest inventories is determining which trees within an inventory are subsampled to receive additional measurements to improve stand level characterizations. Iles (2012) evaluated how the selection and intensity of sample trees including additional observations of height, taper, crown form, and periodic diameter growth in variable radius plot sampling can impact stand

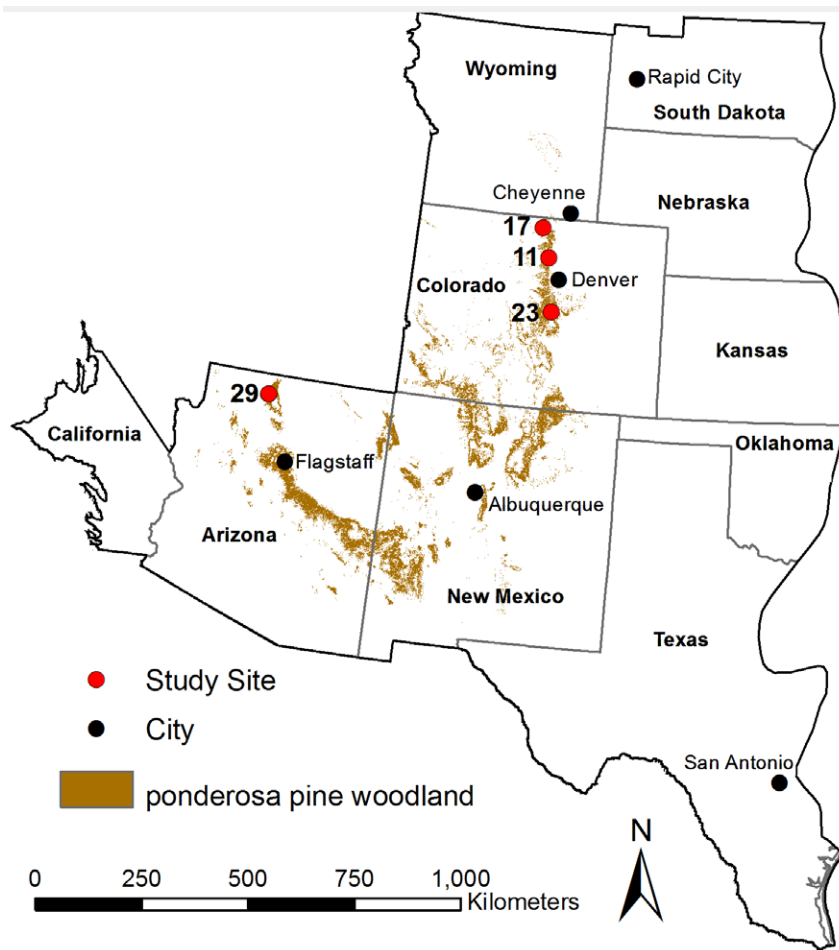


Figure 1—Map of study area showing extent of the Central Rockies Variant of the Forest Vegetation Simulator and the range of the ponderosa pine woodland ecosystem within it. Study site locations are indicated on the map and labeled with their site index at a base age of 100 years.

estimates of timber volume, showing that spatially dispersed sample trees provided the greatest benefit. However, such analysis of sample tree selection intensity has not been conducted to evaluate the estimation of stand characteristics controlling crown fire potential.

This study utilizes four stem mapped ponderosa pine stands at different site indices, inventoried with complete censuses of tree DBH, height, and crown ratios to evaluate the proportion of an inventory needing subsampling of height and crown ratio records to accurately estimate stand level CBH and CBD in FVS. Specifically, we evaluate (1) the tree level error within the dubbing of heights and crown ratios as a function of site index and percentage subsampling of tree height and crown ratio in the inventory, and (2) the percentage of subsampled

tree heights versus the percentage of tree heights with crown ratios needed to achieve a stand level estimate of CBH and CBD within 10 and 20 percent of the census inventory. The results are further discussed for their implications on the selection of subsample trees within management focused inventories.

METHODS

Study Region

Within CR, the ponderosa pine woodland ecosystem occupies much of the managed forested land along the lower treeline ecotone and is generally adjacent to populated areas (Comer and others 2002) (fig. 1). Throughout the region this system is dominated by ponderosa pine with smaller representations

of Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco.], quaking aspen (*Populus tremuloides* Michx.), Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), and blue spruce (*Picea pungens* Engelm.) on more mesic sites and Rocky Mountain juniper (*Juniperus scopulorum* Sarg.) and two-needle pinyon (*Pinus edulis* Engelm.) on more xeric sites. This forest system has seen growing levels of treatment over the last several decades attempting to restore historical spatial pattern and process, among the objectives of these treatments is the restoration of stand structures conducive to a high frequency mixed-severity fire regime (Dickinson 2014). The CR variant contains five sub-models for tree growth that all utilize slightly different coefficients in calculating crown length, to ensure the stands were able to be directly compared to each other, all stands were set to be located on the Arapaho, Roosevelt National Forest which defaults the Lodgepole Pine sub-model.

FVS Simulation Data and Structure

Four stem mapped stands representing site indices for ponderosa pine of 11, 17, 23, or 29 m (base age of 100 years) were census inventoried, with species, DBH, height, and crown ratio recorded on all trees taller than 1.37 m (4.5 feet). These four 4 ha stands were selected for being at similar stages of development and in need of a treatment to restore spatial heterogeneity and to reduce the likelihood of a high severity fire. Each stand was inventoried using an interlocking grid of 64 square 25 by 25 m plots. Prior to inventory, the stands were each subjected to a variable retention

harvest treatment with the objectives of restoring both a spatially heterogeneous pattern and a forest structure believed to be resilient to wildfires. These treatments thinned throughout the diameter distribution range and created an uneven aged forest structure, see Tinkham and others (2016a) for further details on the stands management and inventory (table 1).

To prepare each stand for simulation in CR, tree records within each of the 64 square plots were randomly subsampled for tree height or combinations of tree height and crown ratio records from 0-100 percent in 5 percent increments. Each subsampling increment was replicated 10 times for a total of 383 simulations of each stand. The stands were simulated in CR (version 1675) to determine stand CBH and CBD and the FVS Treelist of individual tree observations could be output. Within FFE-FVS, CBH is defined as the height where crown fuels reach a density of 0.011 kg m⁻³ and CBD is defined as the maximum value in a 4 m running mean of 0.3 m canopy biomass slices (Rebain 2010, Reinhardt and Crookston 2003). Individual tree records of height and crown ratio from the Treelist at different subsampling intensities were compared against the census derived measures for each tree to determine mean error for each site and replicate. Additionally, the accuracy of each simulation for the CBH and CBD were calculated as the percentage departure from the census inventory scenario. From the relative departures of each simulation, linear regressions were generated through all 383 simulations of each stand to determine the subsampling intensity needed

Table 1—FVS derived estimates of stand structure for the census inventory

Site index (m) [‡]	Trees ha ⁻¹	Basal area (m ² ha ⁻¹)	QMD (cm) [‡]	Height (m) [‡]	CBH (m) [‡]	CBD (kg m ⁻³) [‡]	Species proportion ^{*^}
11	296	14.3	24.9	9.1	3.4	0.060	PIPO (99%)
17	148	9.0	27.9	12.8	4.6	0.034	PIPO (97%) PSME (3%)
23	245	15.5	26.9	18.3	5.2	0.025	PIPO (86%) PSME (11%)
29	190	19.8	36.3	21.6	8.2	0.028	PIPO (100%)

[‡] Site Index is calculated at a base age of 100.

[‡] FVS estimates of QMD – quadratic mean diameter; Height – mean height of 40 largest trees per acre; CBH – canopy base height; CBD – canopy bulk density.

^{*} Species proportion was calculated as percentage of total basal area.

[^] PIPO – ponderosa pine; PSME – Douglas-fir.

to achieve a 10 or 20 percent accuracy across the range of site indices.

RESULTS

The analysis of individual tree errors from the dubbing process of heights revealed that with no inventoried heights that there is high variability in the level of error, but that even small levels of subsampling for tree heights or heights with crown ratios provides a substantial improvement (fig. 2). The level of error within the predicted

heights increased with site index, or as the range of potential tree sizes within a stand increased. Given only heights within an inventory, CR dubbing of missing crown ratio values resulted in a substantial underestimation bias for all subsampling levels that increased as site index increased (fig. 2). This crown ratio underestimation bias started at zero for the lowest site index and increased by 1.1 percent per 1 m increase in site index to approximately 21 percent for a site index of 29. However, the addition of crown ratios to the subsampled tree inventory data led to substantial decreases in the level of bias

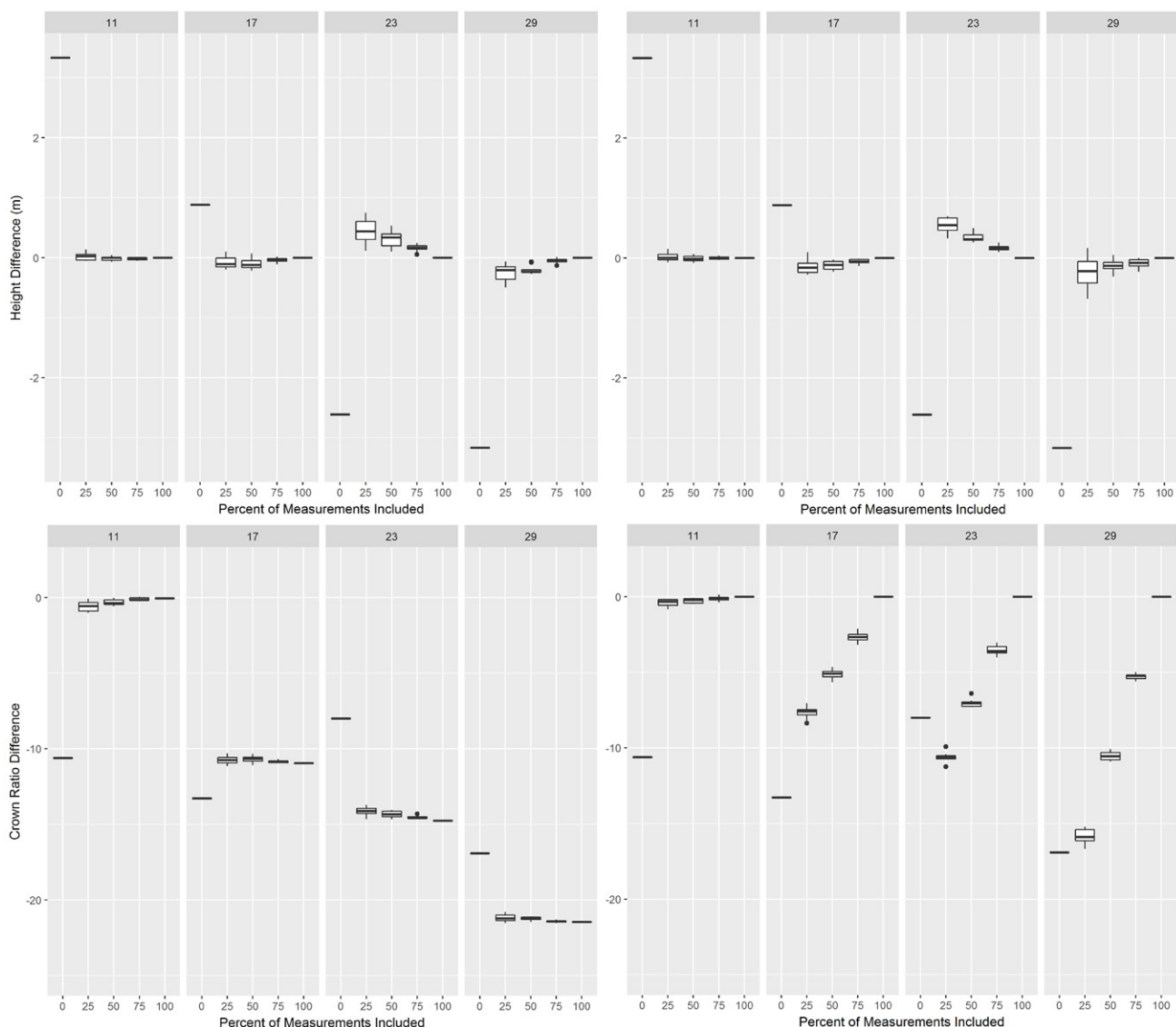


Figure 2—Boxplots of mean individual tree heights (top) and crown ratio (bottom) errors at each site index from the dubbing process at different subsampling intensities of heights (left) and height with crown ratios (right) compared against the census inventory scenario, representing the absolute difference of FVS predicted values from observed. The 25, 50, and 75 percent sampling intensities represent the ten replicated samplings at each site index and metric combination.

(fig. 2). The decrease in bias with subsampling level was most pronounced at higher site indices. There was only a small improvement in crown ratio prediction bias at a site index of 11, however, at a site index of 29 for every 25 percent increase in subsampling trees with crown ratio records there was an approximate 25 percent decrease in the prediction bias.

Looking at the difference between simulations with height and those with height and crown ratio we see that as more subsampled tree observations are added (i.e., as the HR simulations receive a greater level of data), the divergence in their estimated CBD becomes greater (fig. 3). The stand level prediction

of CBD showed generally small errors, but that subsampled trees with only height records tended to have higher relative errors across all site indices and tended to increase in error as the subsampling level increased (fig. 3). To achieve a 20 and 10 percent accuracy threshold for CBD required the inclusion of 17 and 25 percent of inventoried trees to have both height and crown ratio records, respectively (table 2). Conversely, CR stand level predictions of CBH start with an average overestimation of 67 percent when inventories do not include height and crown ratio information, and then sees substantial declines in error as subsampling increases (fig. 4). Inventories with only height information tended to have much higher overestimation errors that never

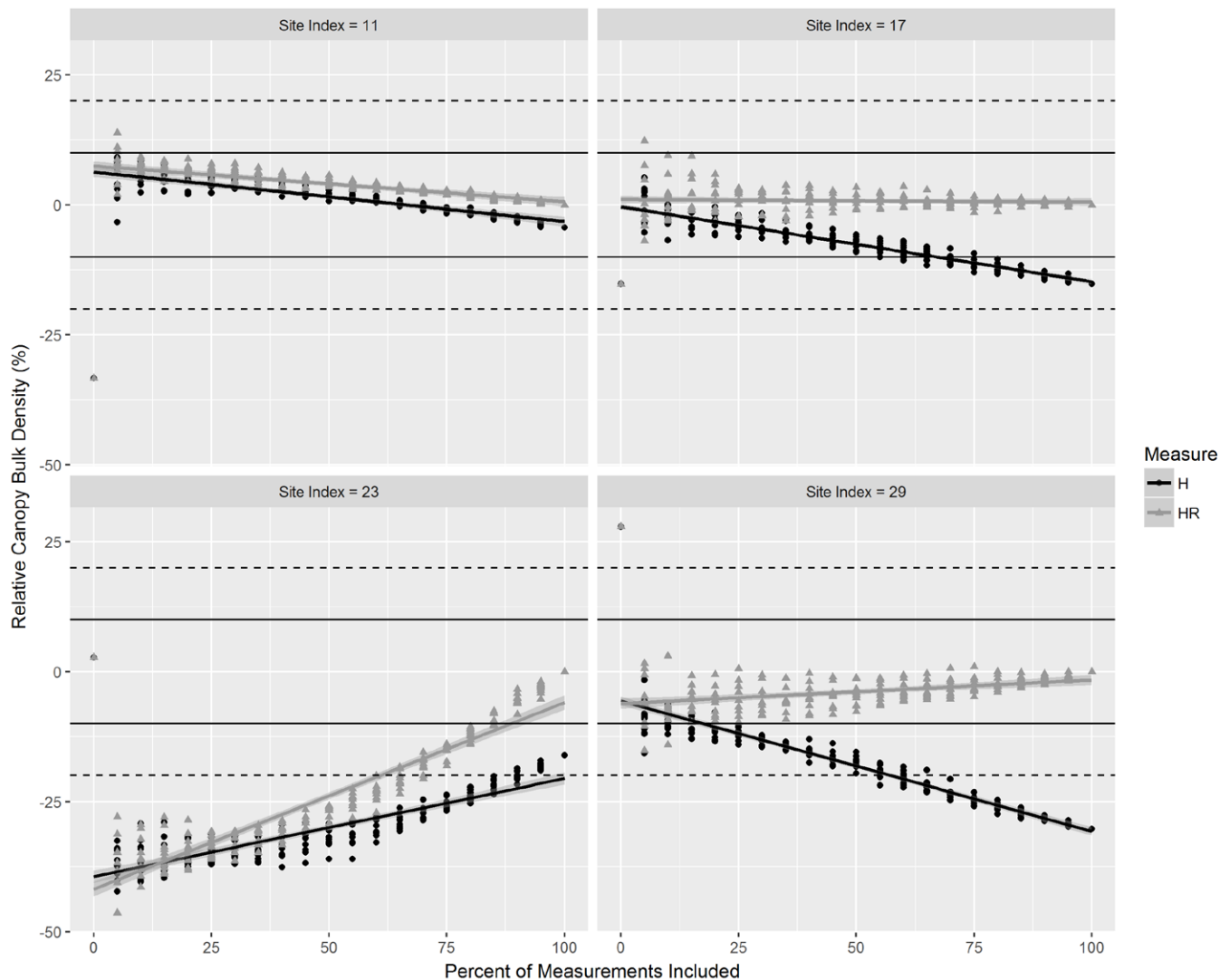


Figure 3—Relative departure of stand level canopy bulk density from the census inventory scenario of each site index for height (H) and height with crown ratio (HR) subsampling simulations ranging from 0 to 100 percent of tree records, with linear regressions through the 383 simulations of each stand. Horizontal lines represent a precision level of 10 percent (solid line) and 20 percent (dashed line) of the census inventory scenario.

Table 2—Linear regression coefficients and predicted subsampling intensity to achieve 10 and 20 percent accuracy, where dashes indicate that the census inventory for that metric could not achieve the accuracy level

Site index	Metric	Canopy bulk density				Canopy base height			
		Intercept	Slope	20% accuracy	10% accuracy	Intercept	Slope	20% accuracy	10% accuracy
11	Height	6.35	-0.10	0	0	25.90	-0.19	31	84
11	Height/Ratio	7.51	-0.07	0	0	22.33	-0.24	10	51
17	Height	-0.33	-0.14	0	72	76.40	-0.04	—	—
17	Height/Ratio	1.04	0.00	0	0	74.98	-0.83	66	78
23	Height	-39.43	0.19	—	—	142.44	-0.67	—	—
23	Height/Ratio	-41.85	0.36	61	89	63.91	-0.72	61	74
29	Height	-5.64	-0.25	57	17	69.22	-0.03	—	—
29	Height/Ratio	-6.16	0.05	0	0	63.77	-0.74	59	72

Zero values mean that the simulations were within the indicated accuracy at a subsampling intensity of 0 percent; a dash means the accuracy cannot be achieved even with a census.

reach the 20 and 10 percent accuracy levels for the three higher site indices. For inventories including both height and crown ratio records, the decline in error happens much more quickly and the 20 and 10 percent accuracies were achieved with subsampling intensities of 49 and 69 percent of inventoried trees (table 2).

DISCUSSION

With no provided height information to allow FVS to calibrate equation 1, there was a positive bias in height predictions for low site indices and negative bias for high site indices. However, for simulations that provided as little as 5 percent of tree records with height measurements so that equation 1 was automatically calibrated to local conditions, there was a substantial improvement in accuracy for individual tree heights (Keyser and Dixon 2008). However, the underestimation bias of the individual tree crown ratio predictions might point to an issue between the models original development and the types of managed stands we are now attempting to simulate. This underestimation bias is similar to the bias seen in evaluation of the FVS North Idaho and Northeast California variants, where similar shifts in stand management have occurred (Leites and others 2009). The data from the GEMGYM project used in developing the relationships in CR included 142 plots dominated by ponderosa pine in both

natural and managed stands collected in the 1980s (Edminster and others 1991). This dataset allowed FVS to represent non-spatially dynamic silvicultural treatments (i.e., thinning from below) common during the data collection of the 1980's, but with an increasing focus on multi-resource management and the restoration of spatial patterns and processes we are asking CR to simulate much more variable and dynamic growing environments. The increasing underestimation bias in CR predicted crown ratios with increasing site index implies that inventories concerned with capturing crown ratio dynamics should be sure to measure this attribute in the field. Additionally, the level of subsampled trees within an inventory including crown ratios may need to be increased as site index increases to ensure accurate modeling within CR. However, for CR to improve in its dubbing of missing crown ratio records an adaptive learning routine like that used in filling missing height records may be the best solution.

The relatively high precision of stand level CBD estimates from CR using the Lodgepole Pine sub-model should be expected since FFE predicts individual tree crown biomass as a function of DBH and then uniformly distributes this across the crown length (Brown and Johnson 1976, Reinhardt and Crookston 2003). However, we can see that the use of height only inventory data can lead to substantial underestimation of CBD

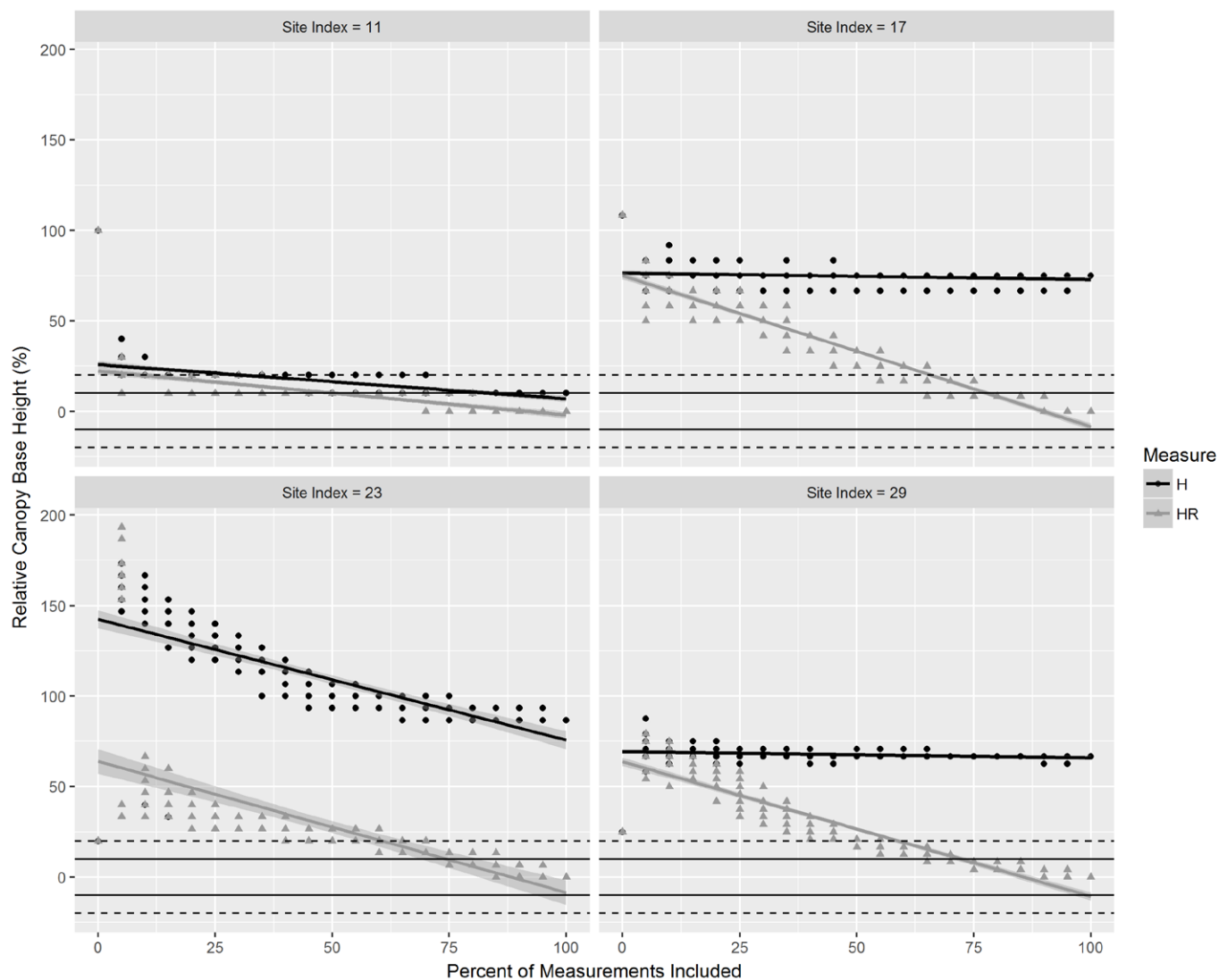


Figure 4—Relative departure of stand level canopy base height from the census inventory scenario of each site index for height (H) and height with crown ratio (HR) subsampling simulations ranging from 0 to 100 percent of tree records, with linear regressions through the 383 simulations of each stand. Horizontal lines represent a precision level of 10 percent (solid line) and 20 percent (dashed line) of the census inventory scenario.

as site index increases, which is related to the underestimation bias of crown ratios at higher site indices (fig. 3). At high site indices, this crown ratio bias would lead to fewer crowns overlapping and result in the decreased CBD that is predicted. The accurate representation of CBD is a key parameter to modeling active crown fire behavior and in recent years research has highlighted the need to incorporate spatially and vertically dynamic distributions of crown fuels to more accurately model these hazards (Cruz and Alexander 2010, Ex and others 2016, Hoffman and others 2016).

Since we see such a strong underestimation bias in the predicted crown ratio values, this propagates through to cause a significant overestimation of CBH (fig. 4). This overestimation happens because the canopy biomass is being distributed into a shorter crown length, leading to fewer crowns overlapping within the canopy and causing FFE-FVS’s critical crown biomass density threshold of 0.011 kg m^{-3} to be reached at a greater height. This increase in CBH is similar to the increase seen by Tinkham and others (2016a) who looked at the temporal propagation of errors in FVS modeling

of crown fire predictors. This overestimation bias of CBH implies the model is less likely to simulate torching of trees within a stand when crown ratio observations are not included.

The influence of sample tree selection and intensity on estimate accuracy has been evaluated for different sampling unit designs looking at estimation of characteristics like timber volume (Henttonen and Kangas 2015). A result of many of these studies demonstrates that distributing sample trees for advanced measurements like height and crown ratio across a population can improve estimates of timber volume (Iles 2012). However, little work has been done to characterize the intensity of sample trees requiring these advanced measurements to adequately model crown fire behavior and hazard. The results of this study show that inventories attempting to characterize these fire management objectives in CR should consider a sample tree selection scheme that balances both height and crown ratio observations. Inventorying for both height and crown ratio on subsample trees led to an approximate 6 percent reduction in the sampling intensity needed to achieve a 20 and 10 percent accuracy for CBD (table 2). This improvement is even greater for CBH, where height only inventories failed to achieve a 20 percent accuracy for three of the sites even at a census inventory. When crown ratio observations are included in inventory data, sample tree intensities of 49 and 68 percent on average are needed to achieve 20 and 10 percent accuracies (table 2). However, if the dubbing process of crown length and subsequently crown ratio were improved by adding an adaptive learning routine, it is believed that similar accuracies could be achieved for CBH as those achieved for CBD.

CONCLUSION

Sample tree selection intensity for the inclusion of either height or height with crown ratio in CR was evaluated for its implications on model prediction accuracy of stand level CBD and CBH. Results indicate that CR provides consistent estimates of CBD for inventories using either height or height with crown ratio records at most sampling intensities. However, model predictions of CBH exhibited a high overestimation bias that could only be corrected by including crown ratio observations

for sample trees. This systematic bias in crown ratio would lead to an underprediction of passive crown fire potential and highlights a need for research to improve the modeling of missing crown structure parameters. Inventories seeking to model crown fire potential with CR should consider the added benefits that crown ratio observations have on model accuracy. Additionally, the inclusion of crown ratio observations in inventory data will have direct impacts on FVS projections of future growth.

ACKNOWLEDGMENTS

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Fire, Carbon, and Climate Projects

Using Climate-FVS to Inform Management Decisions: Three Case Studies from the American Southwest

Andrew Sánchez Meador, Alicia Azpeleta, Michael Stoddard, Benjamin Bagdon, and Sushil Nepal¹

Managers are increasingly being tasked with considering how current and planned actions may affect forests' ability to adapt to a changing climate, yet the toolset for evaluating these types of scenarios is extremely limited. Even more challenging is the fact that predictions and assessments of uncertainty surrounding treatment-climate interactions are poorly understood and not easily obtained, which highlights the need for "climate-aware" decision support tools like Climate-FVS (Crookston and others 2010). We present the results of three case studies, in which Climate-FVS was used to help inform important management decisions. In brief, Climate-FVS allows users to incorporate the potential effects of climate via: (1) functions that link tree mortality and regeneration to climate variables through species viability scores, (2) explicitly linking site index, and thus potential growth, to climate, (3) integration of functionality for changing growth rates due to climate-induced genetic responses and (4) facilitation of tree establishment and potential regeneration shifts (i.e., assisted migration). Species viability scores are obtained for Climate-FVS simulations via a web-based service, which provides climate predictions generated by down-scaling several general circulation model (GCM) outputs and scenarios from the IPCC 3rd and 5th assessments (IPCC 2013).

The first case study presented explored potential forest recovery following uncharacteristic wildfire in Arizona. The Rodeo-Chediski fire of 2002 burned 468,000 acres in Arizona—the largest and most severe wildfire in southwestern U.S. forests to that date. Strom and Fulé (2007) simulated vegetation

change for 100 years following the fire to assess the long-term effects of fuel treatments. Their results suggested that pre-fire treatments affected the forest for over a century, leading to distinctly different vegetation trajectories, whereby untreated ponderosa pine/Gambel oak forest was converted to oak/manzanita shrubfields in a matrix with junipers and New Mexico locust. They concluded that fuel treatments can be valuable for sustaining native forest in the face of uncharacteristic wildfires (Strom and Fulé 2007). Azpeleta and others (2014) also used the Central Rockies variant (Southwestern Ponderosa Pine submodel) of Climate-FVS to compare alternative climate and management scenarios on the Rodeo-Chediski landscape. They incorporated seven combinations of GCM and emissions scenarios to make 100-year predictions of forest conditions compared with an unchanging climate scenario. This no climate scenario predicted a gradual increase in forest density and carbon stocks. In contrast, scenarios that included continuing high levels of greenhouse gas emissions led to near-complete deforestation by 2111. They tested six management strategies aimed at sustaining future forests, which included varying thinning intensities and prescribed burning intervals, and compared these with a no treatment scenario. Severe climate change (estimated increase of 3-9 °F) led to deforestation under all management regimes, but important differences emerged under moderate scenarios (estimated increase of 3-9 °F): treatments that included regular prescribed burning fostered low density, wildfire-resistant forests composed of the current naturally dominant species. When treatments did not include fire under moderate climate change scenarios, forests were forecast to

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become dense and susceptible to severe wildfire, with a shift to dominance of sprouting species.

The second case study examined potential forest trajectories following restoration treatments in dry mixed-conifer forests of southern Colorado. These forests provide important wildlife habitat and supply multiple ecosystem services, including watershed protection, carbon sequestration, and nutrient cycling (Evans and others 2011, Reynolds and others 2013). While most western dry mixed-conifer forests are at risk of uncharacteristically large, high-severity fires, managers can mitigate this hazard over the short term by reducing density and shifting species composition toward fire tolerant species by implementing ecological restoration treatments that utilize mechanical thinning and prescribed fire (Korb and others 2012, 2013). However, empirical information on the long-term impacts of these treatments is limited, especially in light of potential climate change. Stoddard and others (2015) assessed changes in forest structure and composition 5 years following three alternative restoration treatments in a warm/dry mixed-conifer forest on the San Juan National Forest, and used Central Rockies variant (Southwestern Mixed Conifers submodel) of Climate-FVS to model potential forest trajectories under alternative climate scenarios (Representative Concentration Pathways, or RCP 4.5, 6.0, and 8.5). They reported that thinning and prescribed burning treatments were more successful in maintaining resilient forest conditions compared to burn only and control treatments, which led to sparse forest conditions and conversion to sprouting shrubfields.

The last case study focused on assessing potential growth, mortality, and carbon stores on the Four Forest Restoration Initiative (4FRI) landscape of the U.S. Forest Service Southwestern Region. In 2009, Congress established the Collaborative Forest Landscape Restoration Program (CFLRP) (USDA 2015) with the purpose of encouraging collaborative, science-based ecosystem restoration of priority forest landscapes. Largest among the projects funded through CFLRP is 4FRI, a 2.4 million acre restoration project spanning the Apache-Sitgreaves, Coconino, Kaibab and Tonto National Forests of northern Arizona. Bagdon and Huang (2014) used the Central Rockies variant (Southwestern Ponderosa Pine submodel) of

Climate-FVS to examine ponderosa pine stands grouped along an elevational gradient and examined changes in growth, mortality, and carbon stores over a 100-year projection period under three management and three climate scenarios (RCP 4.5, 6.0, and 8.5). Management included a no-action scenario and two scenarios characterized by thinning followed by prescribed burning, each with different intensities and burn intervals, respectively. Results suggested that increases in aridity due to climate warming could result in substantial mortality throughout the elevational range of ponderosa pine, with the most extreme effects predicted to occur in stands at lower elevations. These findings indicate that only the most intensive management scenarios may be effective, in maintaining moderately consistent levels of basal area and increased resilience to climate change compared with the other management scenarios.

While each case study reported slightly different results using Climate-FVS to explore the potential effects and interactions of management, disturbance, and climate, commonalities exist. Each case study reported substantial differences in model outputs depending on climate and management actions, but all suggest that restoration treatments that include both thinning and burning can maintain forest integrity into the foreseeable future. While all three studies suggest that management can improve resiliency to climate change, simulations suggested that for best results, resource managers may need to employ more intensive thinning treatments than have historically been implemented. Furthermore, the case studies presented make a strong argument that, at minimum, managers should incorporate potential climate change effects into the process of evaluating possible alternatives and informing future management decisions.

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Integrating Large Wildfire Simulation and Forest Growth Modeling for Restoration Planning

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Abstract—One of the major science gaps in U.S. wildfire policy is the lack of studies on the long-term benefits of hazardous fuel reduction and restoration programs. For instance, there is little information available to predict the impact of current fuel management and restoration on wildfire activity and whether these fuel reduction activities will meet expectations in terms of wildfire risk to social, ecological, and economic values on national forests. To address this gap, we built a new model that uses the Forest Vegetation Simulator Parallel Processing Extension and FSim large wildfire simulator model to simulate forest management on large landscapes (e.g., 1-5 million ha). We are using the model to analyze 50-year management scenarios where spatial treatment strategies and intensities are varied, and landscape response is measured in terms of future risk and avoided suppression costs. Here we present initial simulations and discuss future application of the model.

INTRODUCTION

One of the major science gaps in U.S. wildfire policy is the lack of studies on the long-term effects of hazardous fuel reduction and restoration programs. For instance, there are very few studies that predict the effects of current fuel management programs and wildfire activity through time on wildfire risk to social, ecological, and economic values on national forests. Similarly, we do not have models to test the efficacy of strategies concerning the increased use of fire for resource benefit in concert with restoration and fuel reduction programs. Part of the problem is that modeling fuel treatments to assess long-term fuel management strategies on large landscapes requires a robust forest modeling platform with the capacity to model the dynamics of fuel treatments, wildfire, and succession for individual stands (e.g., 2-20 ha) at the scale of multiple large wildfire events (e.g., 10⁶ ha). There are relatively few models that have this capacity, and the respective application of these models each used a different set of assumptions and modeling approaches with respect to the various modeling components (Barros and others 2017, Conlisk and others 2015, Finney and others 2006, Loudermilk and others 2014, Millington and others 2008, Scheller and

others 2011, Spies and others 2017, Syphard and others 2011). Modeling realistic fuel treatment scenarios requires simulating mechanical thinning, surface fuels mastication, piling, and prescribed fire, which are sequenced over the span of several years. Silvicultural prescriptions are tailored to individual stands based on ecological departure (Haugo and others 2015), stand structure, species composition, and fuel structure, with the objective of recreating fire resilient forests. Stand treatments must be spatially arranged within planning areas in a way that meets landscape objectives related to fire (protection versus restoration), and landscape treatment unit patterns must be replicated in terms of the size, arrangement, and dimensions of actual fuel treatment projects to correctly represent their effects on fire spread rates and intensities (Finney 2001). Equally important is correct representation of post treatment fire spread rates as well as vegetation and fuels recovery through time. The complexity of the modeling is amplified on typical Western United States landscapes that are mosaics of public, private, and private industrial ownerships, each having respective operational, legal, and economic constraints, and motivations to manage forests and fuels towards particular ecological and socioeconomic goals (Charnley and others 2015).

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Further complicating modeling issues on federal lands in the Western United States is the myriad of forest management plan constraints (Ringo and others 2015) that result in mosaics of ecological and amenity reserves on 50 to 60 percent of the forested area.

Modeling wildfire also has a number of challenges including calibration under different weather conditions and replicating spatial ignition patterns and historical fire size distributions (Finney and others 2011, Salis and others 2016). Large fires (e.g., 20 000 – 100 000 ha) in the Western United States are relatively rare events that have little or no historical precedence at the scale of a typical study area used in landscape fire modeling studies. Uncertainty regarding the effectiveness of fire suppression activities under variable weather and topography also complicates simulations (Finney and others 2009).

To further advance forest landscape simulation modeling we expanded on several previous studies by integrating the FSim large wildfire simulation model (Finney and others 2011) into the Forest Vegetation Simulator Parallel Processing Extension (FVS-PPE) and optimizing the FVS code to simulate large landscapes over time. In contrast to Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) (Rebain 2010), the FSim model simulates the spread and intensity of wildfire events rather than fire behavior and effects in an individual stand. We are now completing case studies in central Oregon, the Blue Mountains in eastern Oregon, and northern Idaho. In each study area we analyze a number of management scenarios in which spatial treatment strategies and intensities are varied and landscape responses are measured over 50-year simulations. Response variables include burned area and severity, wildfire impacts on the wildland urban interface, and a cost-benefit analysis of fuel treatments in terms of suppression costs. Here we describe the model and present initial simulation results as well as discuss future application.

METHODS

Model Overview

The LSim model was built by modifying the Parallel Processing Extension to FVS (FVS-

PPE) (Crookston and Stage 1991) to enhance its capabilities and improve performance in areas specific to modeling wildfire and wildfire effects, then integrating the FSim wildfire simulation system with it. The model was created using FVS-PPE code downloaded in 2012. The original PPE extended FVS to allow a list of stands in a landscape to be processed one cycle at a time and makes FVS outputs available to external processes between FVS cycles. PPE can model dynamic interactions between adjacent stands, and place landscape-level constraints and goals on management activities. The utility of the program for landscape forest simulations was demonstrated in several research papers (Ager and others 2010b, Finney and others 2007). However, PPE had a number of limitations including landscape size (number of stands), processing speed, and outputs. Performance was a particular issue, with simulations of 8,000-9,000 stands (about one-fourth of a national forest) requiring several days to a week to complete.

LSim consists of a modified FVS-PPE that controls the system by calling various other components. Some of these components are available as command line programs, such as FSim (Finney and others 2010) while others were specifically developed for this simulation system and imbedded within LSim. Our modifications were built out of open FVS source code (revision 11/20/13) for FVS-PPE and the Southern Oregon and Northeast California variant of FVS (Keyser 2008). Modifications included: (1) removing the limit on the number of stands to simulate, (2) multi-threading algorithms to use multiple processors, (3) between-cycle data are now stored in RAM, rather than written to text files, (4) custom fuel model selection logic replaced default Fire and Fuels Extension to (FFE) logic, (5) a new prioritizing module that provides for multi-scale prioritization of both stands and planning areas, and (6) miscellaneous code modifications to streamline performance. The modifications made it feasible to simulate 50-year scenarios for 50,000 stands (600 000 ha) in 30 minutes. All internal FVS calculations with respect to growth, mortality, and other aspects of forest dynamics remained unchanged. These modifications make updating the base FVS code less straightforward than simply dropping in the latest version, but as updates

have been made with regards to growth and yield calculations, we have incorporated those directly into the code base.

Integrating Wildfire Simulations

FSim is a widely used fire simulation model developed by Mark Finney at the Rocky Mountain Research Station (Finney and others 2011), and simulates large fire events (i.e., ignition, spread, intensity) in contrast to stand-scale fire behavior modeled in the FFE-FVS (Rebain 2010). FSim was created to simulate large numbers (e.g., 50,000) of hypothetical wildfire seasons to address a range of problems related to fire management policy in the United States. FSim employs the Minimum Time Travel (MTT) algorithm. Rates of fire spread and crown fire initiation are predicted by semi-empirical fire behavior equations (Rothermel 1972, Scott and Reinhardt 2001). FSim predicts daily probability of a fire using logistic regression with historical fire occurrence and Energy Release Component (ERC) as input variables, and fire containment using probability models also based on ERC. Weather data for fire simulations are derived from 20-30 year historical records obtained from Remote Automated Weather Stations (RAWS). The simulation operated on a daily time step and the daily probability of a fire was predicted using logistic regression with recent fire occurrence and ERC as input variables. Once a fire is ignited, the daily weather is generated using the results of a time series analysis of daily RAWS weather data (Finney and others 2011). The time series uses estimates of the seasonal trends, the autocorrelation (dependency of a day's ERC value on previous days), and the daily standard deviation to generate synthetic daily weather streams for each day of simulation. Each fire's growth and behavior were simulated from its ignition day through the remainder of the season, or until containment was achieved as predicted based on recent large fires and their recorded sequence of daily activity (Finney and others 2009). The containment model was developed from an analysis of the daily change in fire size to identify intervals of high and low spread for each fire. The containment probability model was found to be positively related to periods of low fire spread (Finney and others 2009). We assumed random ignition locations for simulated fires (Finney and others 2011). Large fire events within the study area have been primarily caused by lightning, and there are insufficient large fire

incidents to detect spatial patterns if they existed. Fire simulations were performed at 270 x 270 m pixel resolution, a scale that permitted relatively fast simulation times and incorporated important spatial variation in fuel data.

Modeling Management Activities

Formulating a forest-wide restoration scenario on a typical national forest is a complex problem owing to a diversity of forest types, management objectives, and land designations. Our approach used detailed information from existing management programs on the Forest, including stand prescriptions, and a landscape scale priority scheme. The stand prescriptions were multipurpose in that they addressed both wildfire behavior and ecological departure from pre-settlement conditions. Fuel treatment prescriptions consisted of a thinning from below followed by a surface fuel reduction treatment and prescribed fire. The simulated treatment regime was specific to each of the major cover types on the Forest as determined from forest vegetation maps. The thinning from below used a threshold set by either trees per ha, stand density index, or basal area depending on the cover type. Prescribed fire parameters were chosen to replicate typical fall prescribed burning on the Forest. We modeled surface fuel reduction treatments using the FUELMOVE keyword and assumed that 90 percent of fuels between 2.54 cm and 30.48 cm in diameter were removed. The post-treatment stand characteristics in terms of fuels required by the simulation models (canopy base height, canopy height, canopy cover and canopy bulk density) were then compared to untreated characteristics for the same year to determine adjustment factors to represent canopy fuels of treated stands. This latter analysis was performed with FFE-FVS for a sample of 4,194 mapped stands using data from recent stand exams on the Forest. After discussions with local fuels planners we chose a timber-litter (TL2) fuel model (Scott and Burgan 2005) to represent treated stands.

Application

The study area was the 756 634 ha Deschutes National Forest (DNF) in central Oregon and surrounding lands contained within a 4 km buffer. The proclaimed boundary is a smoothed version of the administrative boundary that considers

inholdings as part of the Forest, and thus contained extensive privately owned land (121 000 ha) and WUI (43 000 ha) in addition to the national forest land. The 4 km buffer included lands from adjacent national forests, private land, tribal entities, and the BLM. The physiographic gradients, diversity of vegetation, climate, and management resemble the setting around many national forests throughout the Western United States, and are described in detail elsewhere (Ager and others 2012). The Forest contains extensive stands of lodgepole pine (*Pinus contorta*), ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*) and mountain hemlock (*Tsuga mertensiana*). The Forest has experienced over 8,400 wildland fire ignitions since 1949, mostly caused by lightning during the summer months. Wildfire activity has increased over the past decade with almost 2,000 ignitions and 10 large fire events that combined burned 74 250 ha between 2002 and 2011.

We simulated nine fuel management scenarios that were comprised of three treatment intensities and three priorities. The priorities were: (1) distance to the wildland-urban interface (WUI), (2) stand basal area under 21 inches DBH (BA), and (3) potential volume mortality due to fire (PFMORT). The three treatment intensities treated 7200 ha per year, (the current treatment rate), and twice and three times that rate. The priorities were modeled at both the scale of the planning area and the individual stand. Planning areas were selected based on their overall priority score considering all stands and respective conditions within the particular planning area. For instance, under the potential volume mortality scenario, the planning area with the highest value at that point of time in the simulation was selected for treatment implementation. Treatments were then allocated to eligible stands based on the same prioritization criteria until the treatment threshold was met. We simulated a total of 30 replicates for each scenario using the South Central Oregon Variant. We compared the results for stands that were ineligible for treatment (e.g., wilderness, wild and scenic river recreation areas, research natural areas) and those eligible for treatment based on the DNF Land and Resource Management Plan described in detail in Ager and others (2012).

RESULTS

For space considerations we report here only results for the PFMORT scenario where treatments were prioritized based on potential volume mortality due to fire, estimated within FFE-FVS. Plots of burned area over time for each of the 30 replicates for the PFMORT scenario and the 1X treatment scenario show substantial variability in area burned among years and among replicates (fig. 1). High levels of inter-annual variability reflect historical patterns also shown in figure 1. The high variability in future scenarios underscores the stochastic nature of wildfire in space and time on large fire prone landscapes. Any of the replicate scenarios simulated are equally plausible wildfire futures for our study area and vary widely in terms of the amount and timing of wildfire events. Maps of fire perimeters over time in figure 2 show the spatial distribution of wildfire events for the first and last decades of one selected replicate. Fire perimeters were reasonable facsimiles of historical events within the study area.

Significant temporal trends in area burned were not detectable over the 50-year simulation for the different management intensities, meaning the combined changes in vegetation and fuels from succession and management were not sufficient to change overall fire activity within the study area for any of the three treatment levels (fig. 3). These results were obtained assuming weather consistent with historical patterns in the study area. Area burned for the treated areas (fig. 3B) did decline for the 3X treatment scenario in the initial years of the simulation, but then increased to levels equal to the 1X treatment. Area burned was slightly less for untreatable areas (fig. 3A), primarily because these areas are in higher elevation forests with long fire return intervals compared to the treatable areas. Although the outputs suggested some treatment effects and temporal trends, these differences were minor compared to variability among the replicates.

Average standing merchantable volume killed by wildfire increased over time in the untreatable areas (fig. 4A). In treatable areas volume killed by fire on a per hectare basis was more or less constant with a slight increase in year 2040. The results underscored the importance of measuring the effect of fuel management on wildfire behavior within areas that can be treated versus at the scale of national forests, where on average about 50 percent of the land

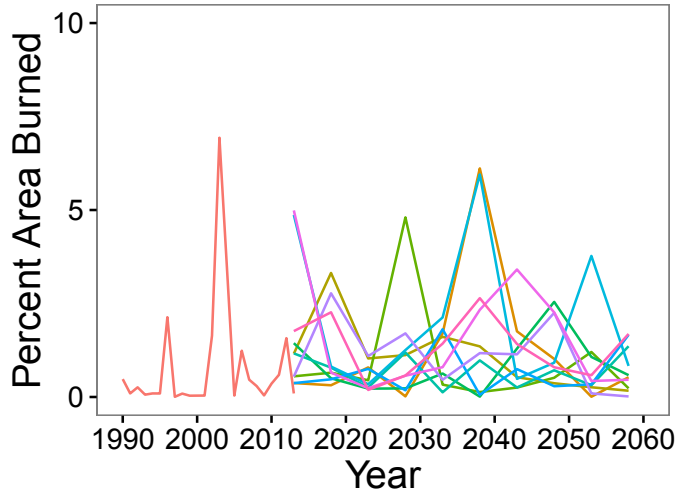


Figure 1—Area burned among 10 replicates for the scenario where treatments were prioritized based on potential fire mortality (PFMORT) under the mid-range treatment intensity (14 400 ha year⁻¹). The historical area burned is included for the same area from 1990-2012. Graph shows variability among future wildfire scenarios associated with replicate simulations.

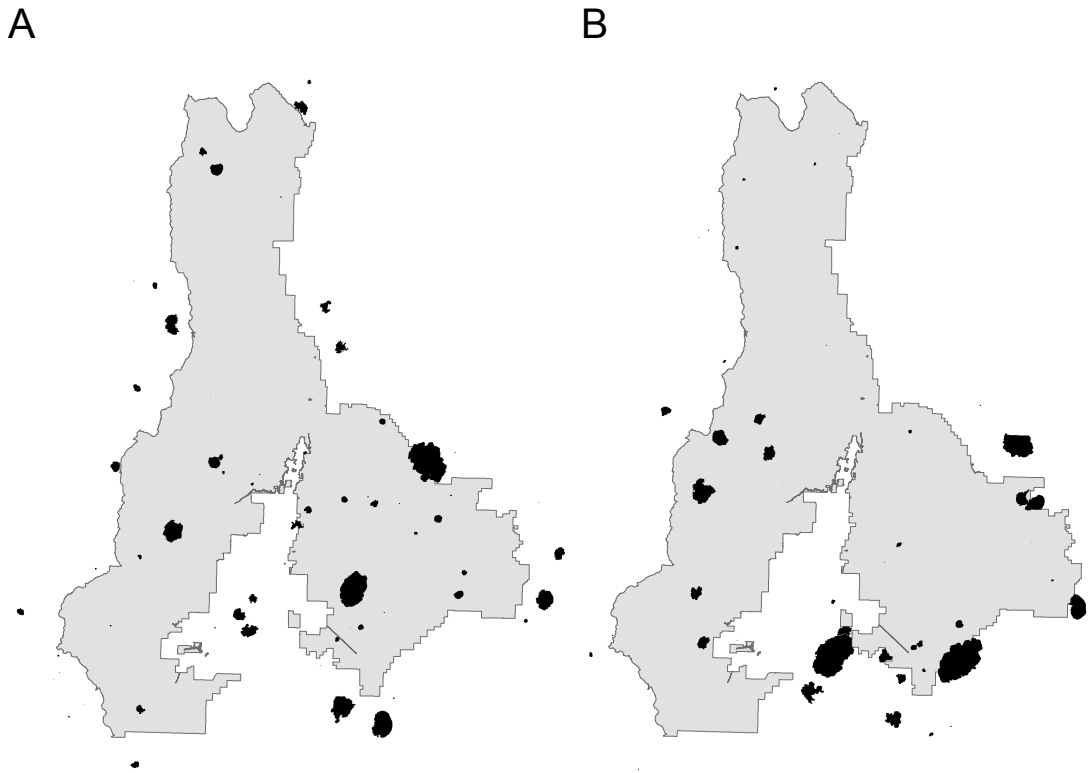


Figure 2—Fire perimeters for a single replicate for the scenario where treatments were prioritized based on potential fire mortality (PFMORT), (A) decade 1, and (B) decade 5 showing spatial variability in fire locations during the simulation.

cannot be treated due to forest planning and other legislated restrictions.

DISCUSSION

This work helps fill a gap in strategic restoration planning by providing a platform to help managers understand the long-term dynamics of forests, restoration policies, fuels, management scenarios and fire. Despite the large budget for field treatment programs in the National Forest System [\$358 million per year (USDA Forest Service 2014)], and the extensive area treated [>1 million ha per year in FY2013 (USDA Forest Service 2014)], decision support tools to understand the landscape-scale effectiveness of fuel treatment programs and their synergistic effect on succession over the long term

do not exist. This modeling system can be used to test the long-term effectiveness of accelerated restoration policies and programs to build fire resilient landscapes on national forests. The fine spatiotemporal scale of the modeling system provides a robust and high resolution platform to analyze fuel treatment strategies on landscapes that are highly fragmented and variable with respect to constraints on mechanical treatments, vegetation, fuels, ownership and weather. In particular, we advanced forest landscape succession and disturbance modeling by integrating a widely applied mechanistic wildfire simulation system with a forest growth simulator that has been calibrated for a wide range of forest ecosystems. The fire modeling system builds an important bridge between forest planning efforts on national forests

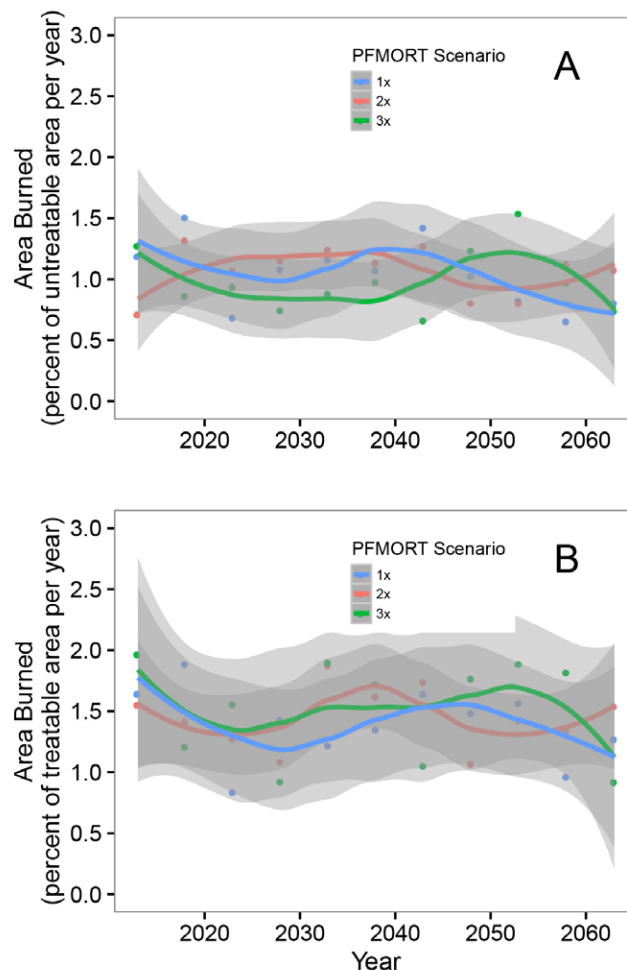


Figure 3—Average annual area burned for the scenario where treatments were prioritized based on potential fire mortality (PFMORT). (A) stands ineligible for treatment (e.g., wilderness); (B) stands that are eligible for treatment based on the Deschutes National Forest Plan. Data are averaged over 30 replicate simulations.

and the fire management programs that use the FlamMap fire behavior library.

FVS-PPE has been used in several previous studies, but none of these incorporated wildfire as an endogenous process within the simulation system. In a previous study in eastern Oregon, the PPE was used to model spatial fuel treatment scenarios that targeted either restoration of upland forest or crown fire in and around urban interface (Ager and others 2010b). In another study the PPE was used to analyze landscape carbon budgets from fuel management (Ager and others 2010a). The PPE was also used by Finney and others (2007) in a detailed temporal landscape modeling study of fuel treatment optimization, but that study did not incorporate wildfire as an endogenous process within the simulation system.

The FSim model and underlying FlamMap code library are widely used for strategic fuels planning and risk assessment in the United States (Thompson and others 2011). The MTT algorithm and associated wrappers are a core component of United States wildfire planning systems (Ager and others 2014, Ager and others 2011, Andrews 2007, Finney and others 2011, Noonan-Wright and others 2011, Rollins 2009, Scott and Burgan 2005) and are used globally in other fire prone systems as well (Alcasena and others 2015, Kalabokidis and others 2015, Oliveira and others 2016, Salis and others 2014). Thus as part of this work we leverage the long history of fire model development in United States federal land management agencies (Systems for Environmental Management 2017).

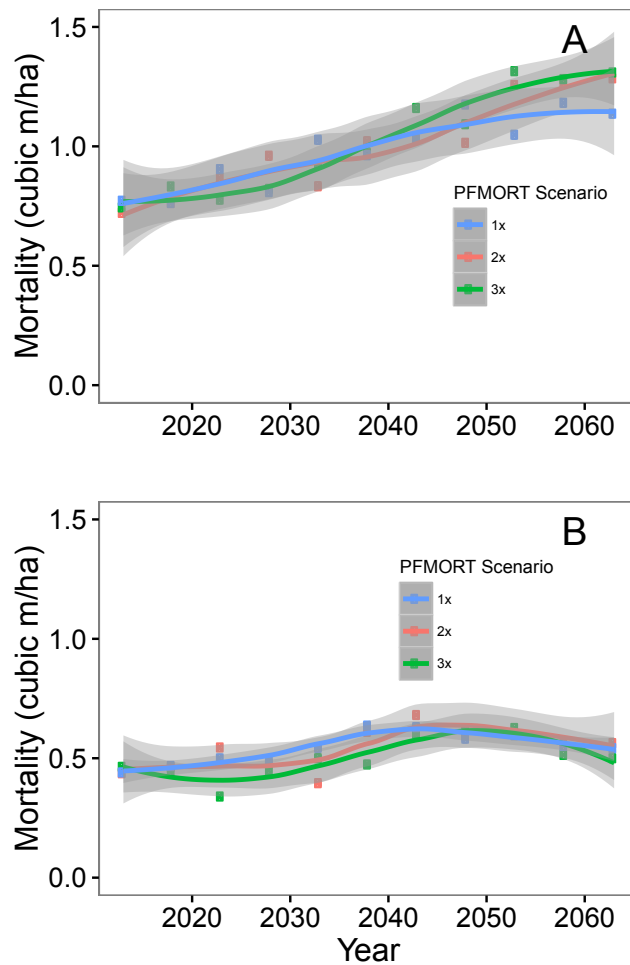


Figure 4—Stand mortality from wildfire. (A) stands ineligible for treatment (e.g., wilderness); (B) stands that are eligible for treatment based on the Deschutes National Forest Plan. Data are averaged over 30 replicate simulations.

The focus of the current paper was describing methodologies for building LSim. Our simulation experiment was primarily conducted to demonstrate the system in concert with the modeling methods and wildfire prediction system. Additional simulation studies will be reported in later communications. The simulations we presented suggest that under assumptions of constant climate wildfire trends under current levels of management are stable. Substantial successional induced changes in surface and canopy fuels are not predicted for the study landscape. This suggests that the system is not at a specific tipping point with respect to fuels accumulation and that the current rate of treating fuels as part of ecological restoration programs (Buford and others 2015, Noss and others 2006) is about the same as fuels accretion. Analysis of variability among years for future wildfire scenarios suggested that extreme fire behavior may or may not be realized in the near term future (e.g., 1–10 years). High variation among years (and replicates), where each represent an alternative future scenario, suggests that management policies may or may not be perceived in the short run as making a significant difference in fire activity. This variability has manifold effects on policy implementation by obscuring trends in wildfire activity in response to restoration and protection programs, and further complicating the assessment of restoration programs and their potential benefits.

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Forest Health Projects

Use of the Forest Vegetation Simulator and the Southern Pine Beetle Event Monitor to Identify Silvicultural Treatments for the Reduction of Southern Pine Beetle Hazard and Enhancement of Restoration on the North Carolina Piedmont

Jason A. Rodrigue, Chad. E. Keyser, and John T. Nowak¹

Abstract—Southern pine beetle (*Dendroctonus frontalis*) is a common disturbance agent in southern pine forests. Forest damage from this insect can be minimized through proper silviculture prescriptions that focus on maintaining vigorous stand growth and reduced understory competition. In this project, we examined approximately 3,000 acres of forest land on the Uwharrie National Forest in the central North Carolina piedmont for hazard to southern pine beetle using a data driven management system called the field sampled vegetation program (FSVeg). Up-to-date common stand inventory information was loaded into the field sampled vegetation program and pulled directly into the Southern Variant (SN) of the Forest Vegetation Simulator (FVS) for calculation of southern pine beetle hazard and other post processed stand level descriptive statistics. Draft silvicultural decisions were considered based on modeled conditions. FVS was then used to model treatments, their outputs, and the effect on stand level southern pine beetle hazard. Based on results, treatments were prioritized and integrated with forest restoration opportunities outlined in the Uwharrie Forest Management Plan. A subset of highest priority treatments were then field validated and made available to move forward into the environmental analysis potentially under the 2014 Farm Bill Categorical Exclusions. This process resulted in several efficiencies that led to rapid development of a decision for treatment under these new authorities.

BACKGROUND

Management on National Forest Lands

Actions to address forest health hazards and other issues requiring management decisions on lands within the National Forest System are orchestrated by a distinct planning to implementation process. Each national forest is guided by a land and resource management plan (Forest Plan) that is based on the National Planning Rule (USDA FS 2012a). All management decisions must move the current land conditions towards a desired set of future conditions outlined in the Forest Plan. Forest Plans also contain objectives that describe distinct metrics to measure the success of implemented actions. Additionally, the effects of management decisions on the environment must be disclosed to the public, public input must be solicited and incorporated, and alternatives considered according to the National Environmental Policy Act of 1969 (NEPA).

Project Development

For the Uwharrie National Forest and the project discussed in this paper, the Forest Plan contained two desired conditions and several objectives that framed the worked proposed (USDA FS 2012b):

Goal 1: Resilience of forests to disturbance. This goal includes restoration of historic fire regimes and reducing threats from nonnative and native pests.

Objective 1: Maintain existing longleaf pine (*Pinus palustris*) forests.

Objective 2: Thin an average of 400 acres a year to “maintain room for growth and to discourage insect and disease infestation.”

Goal 2: Restoration of native pine communities. This includes oak (*Quercus* spp.)/shortleaf pine (*Pinus echinata*) and longleaf pine. The emphasis on this restoration is placed on matching historic communities to site conditions. An additional

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component included controlling the presence of mesic hardwood species such as sweetgum (*Liquidambar styraciflua*) and red maple (*Acer rubrum*) that have developed under an altered disturbance regime (McEwan and others 2011) which includes the absence of landscape level fire (Nowacki and Abrams 2008).

Objective 3: Begin restoration of site-appropriate vegetation each year on an average of at least 200 acres of potential oak/hickory (*Carya* spp.) sites and 100 acres of potential longleaf pine sites.

Disclosing of Effects

At the time of its development insect and disease treatment categorical exclusions was thought to be an efficient vehicle to guide this project. The 2014 Farm Bill amended the Healthy Forests Restoration Act of 2003 to include insect and disease treatment areas on National Forest System lands. Designated by the Chief of the U.S. Forest Service and recommended through each State's Governor's office, projects within these lands that reduce the risk, extent of, or increase the resilience to, insect or disease infestations can use expedited NEPA reviews (categorical exclusions). These projects cannot occur in wilderness, be larger than 3,000 acres, or build new permanent roads. They must: (1) maximize old growth and large trees to the extent the trees promote stands that are resilient to insect and disease conditions, (2) use best available science, and (3) be developed collaboratively.

The designation process on the Uwharrie National Forest identified five watersheds that had high potential for basal area loss from southern pine beetle (*Dendroctonus frontalis*) (SPB). This potential loss was based on Forest Health Protection models (Krist and others 2014).

Within this framework the project moved forward. Along with lowering hazard to SPB, activities would increase the health, growth, and resilience of treated stands. The prescriptions for treatment would accentuate longleaf pine and mixed oak/shortleaf ecosystems, restore their historic structure and composition, and reintroduce fire. Many stands were found to have undesirable understory and midstory hardwood compositions. This condition is

problematic from SPB, ecological, and prescribed fire standpoints.

METHODS

The Uwharrie National Forest

The Uwharrie National Forest is one of four national forests in North Carolina. Located northeast of the city of Charlotte, the forest consists of approximately 60 fragmented parcels totaling 50,000 acres. The Uwharrie was purchased in the 1930s. Past land use including intensive farming and industrial forest company ownership has resulted in forests dominated by loblolly pine (*Pinus taeda*) plantations and shortleaf pine-dominated stands. Elevations range from 400 to 1000 feet above sea level.

Southern Pine Beetle Biology

The southern pine beetle is a native bark beetle in the forests of the Southeastern United States. Its full range extends from southern New York to Central America (Clarke and Nowak 2009). Though a natural part of southern pine forests, SPB has the ability to cyclically build to population levels that can mass attack and kill healthy southern pine tree species. The last SPB outbreak in the North Carolina piedmont was the early to mid-2000s (Nowak and others 2016). All southern pines are susceptible to SPB attack. In the piedmont of North Carolina, pine species considered moderately susceptible to SPB attack include pitch pine (*Pinus rigida*), longleaf pine, and white pine (*Pinus strobus*). Moderately susceptible species tend to have greater amounts of sap flow allowing them to "pitch out" during endemic SPB population levels (Hodges and others 1979, Sullivan 2011). Pine species considered highly susceptible to SPB attack include loblolly pine, shortleaf pine, and Virginia pine (*Pinus virginiana*) (Clarke and Nowak 2009).

Southern Pine Beetle Hazard Management

Stand density is thought to be one of the most critical factors in SPB spot initiation and expansion (Nebeker and Hodges 1983). Lower quality sites (i.e., low nutrient availability and xeric sites) may be considered contributing factors to tree stress, increasing the likelihood of successful SPB attack.

Management to reduce SPB hazard focuses on three goals:

1. To increase availability of site resources (sunlight, water, and nutrients);
2. To change microsite conditions (increase inter/intra-stand air flow); and
3. To increase the growth and vigor of residual trees (greater crown volume, increased root growth, reduce inter-tree competition).

Management actions typically include:

1. Reducing overstory density through thinning, focusing resources on remaining individuals. Thinning also has the ancillary benefit of capturing density dependent mortality before it occurs. The most commonly accepted thinning recommendation involves thinning pine stands with basal areas > 120 square feet per acre to < 80 square feet per acre (Nebeker and Hodges 1983, Nowak and others 2008).
2. Modifying midstory conditions through burning or mechanical and chemical means. Midstory reduction, when combined with overstory thinning, increases air movement within the stand dispersing, disrupting, or diluting SPB aggregation pheromones (Thistle and others 2004, 2011).
3. Ensuring the dominant overstory pine species is best suited for the site. On the Uwharrie National Forest, loblolly pine was planted across the landscape during industrial company ownership. Commonly more suited for mesic sites, loblolly when found on higher slope positions is likely on a site more suited for longleaf pine.

Data Collection

Stand inventories on 3,400 acres of National Forest lands were gathered via contract. Four hundred Common Stand Exam (CSE) Quick Plots were taken with an average acreage per plot of 8 acres. This plot average is consistent with plot intensities recommended by the field sampled vegetation program (FSVeg) CSE protocols for forests with a mix of homogenous (plantation) and heterogeneous (natural) stand conditions (USDA FS 2015). The contractor was provided a list of stands, the number

of plots per stand, and maps. Stands included were selected based on local knowledge, land use history, known vegetation conditions, and whether access was possible. Digitally collected plot data was loaded into the national FSVeg database.

FVS Model Development

Stand data in FSVeg was downloaded into Forest Vegetation Simulator (FVS) formatted files using the FSVeg FVS_DB_LINK utility. Ecoregion was set at the subsection level (Cleland and others 2007), and average stand age was computed based on measurements from growth sample trees. The southern variant (SN) of FVS, version 1778, was chosen, and a 20-year simulation with 3-year cycles was selected (Dixon 2002, Keyser 2008).

The estimation of SPB hazard was generated using the southern pine beetle event monitor addfile. The SPB event monitor addfile was developed by the Forest Health Technology Enterprise Team and SPB specialists (Courter 2002). Out of the three versions available (coastal, piedmont, and mountain), the coastal version was employed with this model because it contained longleaf pine, an important species for management on the Uwharrie that was not present in the piedmont version. The addfile was used to estimate stand susceptibility (hazard) based on the predicted amount of basal area loss due to potential SPB activity. Low hazard ranged from 0- to 20-percent loss, moderate hazard ranged from 20- 40-percent loss, and high hazard was > 40-percent loss.

RESULTS AND DISCUSSION

FVS Model Outputs

All stands in the project were run through SN to develop a database of current conditions (2016) and modeled future stand conditions (2036). This allowed changes in key descriptive characteristics including SPB hazard to be considered in preliminary prescription development. Data from two FVS Database Extension output tables were utilized and merged into a single stand table using the Stand ID attribute (table 1):

1. The FVS_Summary table is the standard output table from any FVS simulation. It contains basic stand summary information such as stand age,

Table 1—Main outputs from the FVS_Summary and FVS_Compute tables used to develop preliminary prescriptions in the Uwharrie Southern Pine Beetle Project

FVS_Summary Table	FVS_Compute Table
Stand Age (years)	SPB hazard rating
Trees per acre (TPA)	SPB hazard rating change
Basal area per acre (sqft/acre)	Pine Species BA (individual species)
Quadratic mean diameter (QMD (inches))	Percent pine in stand (Basal Area basis)
Stand volume calculations (CCF per acre)	Total pine Basal Area (sqft/acre)
Mean annual increment	Stand height (feet)
Stand ID	Stand ID

basal area per acre, stand volume estimates, and quadratic mean diameter.

- The FVS_Compute table contains custom output variables (event monitor calculations) in a table. It included information from the SPB event monitor including SPB hazard ratings, basal areas specific to individual pine species, and percent pine in the stand.

A portion of the stands in the project are routinely burned on a 3-year return interval within the district prescribed fire program. Nowak and others (2015) found the existence of prescribed fire to have some effect on within stand SPB dynamics. Consequently, the FVS model was modified to include prescribed fire effects for those stands identified, and updated modeled vegetation conditions were incorporated into the stand data table. Changes in stand SPB hazard ratings over the simulation period were mapped using Geographic Information System software (ESRI ARC MAP 10.3) to spatially identify conditions within the project area and their relationship to other stands and assess potential access.

Draft Treatment Prescription Development

Preliminary stand treatment prescriptions based on modeled SPB hazard, species composition, basal area, and stand age were developed. A filter identifying stands that did not require treatment was applied first. For example, stands with a SPB hazard rating of 1 in 2016 that also rated a 1 in 2036 were considered for deferred treatment (1 = low risk/hazard). When investigated further, these stands contained a higher proportion of hardwoods, currently contained lower stand basal areas, or were dominated by longleaf pine, which contains

the lowest concern with regards to SPB (Clarke and Nowak 2009). In certain cases, the modeled prescribed burns helped to maintain lower hazard over the simulation period.

The remaining stands were examined for potential overstory treatment (thin or regenerate). The 2014 Farm Bill includes the requirement: “projects will maximize retention of old growth and large trees, as appropriate for the forest type, to the extent that the trees promote stands that are resilient to insect and disease.” For the Uwharrie SPB project, regeneration was still considered an option because mature longleaf pine will be retained in most cases per Forest Plan objectives. Stands of loblolly and shortleaf (though usually in mixed stands) have such a high SPB hazard especially when on longleaf site types that conversion to longleaf communities was warranted. Stands in this latter condition tended to be mature with high basal areas, declining mean annual increment, or poor stand growth.

To better inform those stands being considered for regeneration, a natural vegetation (NV) model was added to the draft treatment development process. The model identifies units of land that support a specific plant community (or group) based upon environmental and physical factors that control vegetation distribution (Simon and others 2005). Potential treatment stands were assessed for NV modeled conditions to further support the decision to regenerate rather than thin.

Information pertaining to past thinning history, prescribed fire, or cultural treatments provides insight for current treatment options. Therefore, the Forest Service activity tracking system (FACTS)

was examined for each stand to determine the past treatment history.

Stand age was an important contributing factor in developing the preliminary prescription(s). A proportion of the project's stands were advanced in age (50 to 110), lessening the benefit of thinning from a SPB hazard standpoint because older trees have a more limited response to thinning (Hicks 1980, Smith and others 1997). Conversely, current research identifies stands under 40 years that have not been thinned as having a high likelihood of outbreak in the presence of the SPB (Nowak and others 2015). Younger stands with high basal areas, even though they were sub-merchantable, also remained in consideration for treatment.

Another decision point was provided by FVS (estimated treatment volumes from the FVS summary table in CCF per acre). FVS estimated stand volumes were compared against historical harvest yields from past Uwharrie timber sale records to determine if individual stands were economically viable as a standalone treatment, required consideration as a non-commercial treatment, or needed to be packaged with higher volume treatments to ensure treatment completion. Value is also important when considering the need to implement cultural treatments like site preparation, prescribed burning, or tree planting within the project as well to ensure movement towards desired conditions.

FVS also allowed preliminary treatments to incorporate local knowledge about thinning thresholds. Removing more than 30 to 50 percent of a stand's basal area in a single entry, depending on the thinning being a first or second thinning, has the potential to dramatically increase the stand's susceptibility to windthrow or put excessive stress (thinning shock) on residuals. This is counterproductive to the objective of reducing SPB hazard. Consequently, some stands could not be thinned to the "below 80 basal area per acre" that is recommended by the forest health community (Nebeker and Hodges 1983, Nowak and others 2008).

Modeling the preliminary prescriptions also provided the post-treatment SPB hazard projection produced by the event monitor providing the project

with further support for completing the treatments (fig. 1). FVS runs that contained potential treatment option effects in combination with changes to SPB hazard, past treatment history, and NV community resulted in draft preliminary stand prescriptions. It was time to go to the field.

Field Verification

Many silviculturists will feel uncomfortable with finalizing recommended stand treatments without first confirming conditions in the field. Field verification to match stand conditions with the draft preliminary prescription is an important next step.

The inventory and FVS modeling work completed prior to stand visits resulted in several efficiencies:

1. Stands with low current and future hazard were prioritized at the bottom of the visitation schedule or did not need a visit at all. This process reduced the acres needing coverage from 3,400 acres to 1,500 acres.
2. Stands whose data did not fully support the preliminary prescriptions or stands that had several options for treatment could be prioritized and questions pre-formed, focusing the stand visit. We were able to focus our time on those stands, finalizing the optimum prescription.
3. Stands whose data and preliminary prescription lined up strongly could have abbreviated visits that confirmed conditions or sought answers to specific questions. Other team members were asked to visit these stands distributing the volume of work.
4. A draft prescription field sheet was completed for each stand containing treatment options, specifics about the stand based on the inventory and model data, and relevant questions and concerns about the specific stand that needed to be verified in the field in order to support moving forward with treatment. Sample questions dealt with the distribution of key species within the stand, whether there is access for mechanical treatment, or if there is evidence of previous treatments. The field sheet also contained the silvicultural intent behind the draft prescription which described the future desired conditions that the treatment

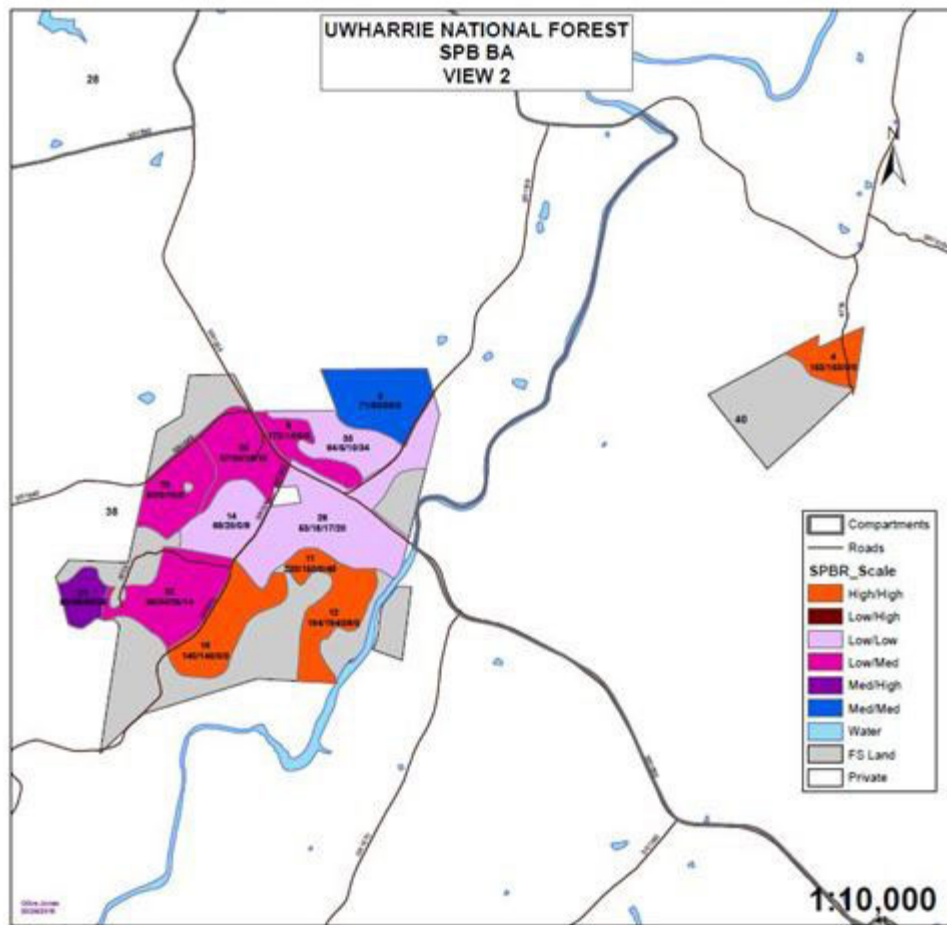


Figure 1—A subset of the Uwharrie National Forest Southern Pine Beetle (SPB) Project area (area 2) characterizing FVS-modeled SPB hazard changes over the 30-year simulation period. SPBR_Scale is defined as the change in modeled SPB hazard from the beginning of the simulation (2016) to the end of the simulation (2036).

was working toward. It also included a spot for check plots to confirm inventory data specifics and other field related notes regarding conditions apparent in the field that are not readily observed in the plot or modeled data (e.g., midstory conditions).

5. Field verification also provided us the opportunity to meet with district operations staff to gain local knowledge about the stands, accessing them with needed equipment, and to discuss access options which may directly impact the actions selected to implement the silvicultural prescription.

The field work for this project was completed over the course of two days. The 1,500 acres were evaluated by a crew of five to six people visiting various stands depending on the needs listed above.

Final prescriptions were completed on 1,100 acres and involved the following changes.

1. Several stands that were thinned previously (1st thinning) contained residual pine spacing with intermixed hardwoods that effectively reduced the SPB hazard. Pursuing treatment under this project would not meet the objective of reducing southern pine beetle hazard. Treatment was deferred.
2. Overstory treatments were altered or switched between regeneration and thinning. If too much of the surrounding landscape was already in young age classes due to previous regeneration treatments, further regeneration was deferred to remain consistent with the Forest Plan and the National Forest Management Act (NFMA). Conversely, field observations of a loblolly

plantation proposed for thinning may have determined it is on a longleaf site or surrounded by other loblolly plantations on private lands, warranting conversion to longleaf through regeneration.

3. Due to constraints in the 2014 Farm Bill Categorical Exclusion language, treatments were dropped because access was not readily available. Alternatively, these stands may be treated non-commercially but the large volume of merchantable sized wood left in the stand may itself be a forest health hazard.
4. Thinning intensities were adjusted. A stand proposed for thinning may contain too many residuals and be at such a high initial stem density that it cannot be operationally implemented with mechanized equipment without introducing high levels of residual stand damage. Considerations were made for thinning to lower densities where those density reduction levels would not harm the residual stand.
5. Many stands currently under a prescribed burn rotation contained midstories heavily stocked with sweetgum and red maple. This condition increased the SPB hazard due to intense competition for resources, decreased airflow microsite conditions, and changes in prescribed fire behavior. Treatments to address these conditions mechanically and chemically were added to several prescriptions.
6. Field observations allowed for consideration of forest conditions and property ownership surrounding stands with draft prescriptions. In one case, adding treatment to surrounding stands would facilitate the addition of the entire Forest Service ownership block to the district prescribed burn rotation facilitating long-term maintenance of lower SPB hazard conditions and small scale landscape restoration potential.

Finalized Treatment Prescriptions

Where the preliminary prescription was changed, a stand was added, or a treatment was dropped, and FVS was re-run to produce modeled results that were consistent with the prescriptions moving forward in the project. This step in the project provided several benefits:

1. A clear picture of SPB hazard;
2. Updated volume estimates;
3. Updated stand conditions;
4. Treatment visuals with SVS (Stand Visualization System); and
5. Materials for work with the public and/or collaborative groups.

Post field observations and finalized prescriptions were packaged to inform the project's movement forward towards environmental review and implementation. Furthermore, detailed silvicultural information was prepared when the picture of stand treatment was fresh and can be passed forward to implementation with the highest degree of accuracy.

The final package consisted of:

1. Relevant maps;
2. A tabular list of stands, treatment suite, acres treated, objectives met, measured and modeled metrics, and relevant stand specific notes; and
3. A detailed prescription sheet identifying silvicultural intent and sequence and timing of associated treatments.

The goal was to prepare the decisionmaker with the tools needed to engage in collaboration with the public and navigate the NEPA process efficiently and effectively. NEPA required disclosure and analysis of effects of proposed actions on the surrounding environment. Having modeled changes to conditions using FVS was a great benefit including increasing the power of the analysis, being well grounded in the best available science, and having a support system built around a nationally supported application. With the FVS model, our ability to gain the support of our collaborators was increased as we can better demonstrate our management intentions.

CONCLUSIONS

In summary, use of updated stand condition data and FVS benefitted the Uwharrie Southern Pine Beetle Project in several ways:

1. It made additional information available to assist with treatment prescription decisions

(i.e., SPB hazard rating changes and estimated volumes) and to the decisionmaker, district staff, public, and collaborative groups.

2. It increased the efficiency of the field portion of the project and allowed us to develop, based on current data, solid initial prescriptions before leaving for the field.
3. It allowed us to overcome a limitation in manpower and time, shortening the field work period, and facilitated prescriptions to be completed remotely.
4. By using FVS to support our silvicultural decisions, it increased the efficiency of informing collaborative groups and completing the NEPA process.

In this paper's example, inventory and FVS was used in conjunction with the 2014 Farm Bill to make silvicultural recommendations on a portion of the Uwharrie National Forest. The authors believe that the same template may be used under a wider array of National Forest District projects to realize similar efficiencies.

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Estimating Changes to Forest Structure as a Result of Forest Pests: Using FVS to Simulate Potential Effects of Emerald Ash Borer Across a Broad Landscape

Andrew J. McMahan and William B. Monahan¹

This study follows the 2013–2027 *National Insect and Disease Forest Risk Assessment* (Krist and others 2014), which assessed nationwide potential future (15-year) mortality to individual tree species resulting from forest insects and diseases. In this project, we assessed possible pest-induced changes to community composition and structural attributes as characterized by forest type (Arner and others 2001), structural stage (Crookston and Stage 1999), and size class. Specifically, we used the Forest Vegetation Simulator (FVS) to estimate potential changes to plant community structure resulting from tree mortality caused by emerald ash borer (*Agrilus planipennis*, EAB) across a large portion of the upper Midwest.

We simulated mortality to ash trees (*Fraxinus* spp.) emulating EAB activity in thousands of treelists representing Forest Inventory and Analysis (FIA) plot inventories from seven states: MN, MI, WI, IN, IL, OH, and PA. Treelists came from LANDFIRE's database of FVS-ready, publicly-available, FIA-sourced inventories. Using true FIA plot locations (under Memorandum Of Understanding with FIA), we identified all treelists (~14,000) originating in EAB counties of the aforementioned States. From that subset, we extracted all treelists containing any ash tree records (~4500) to use for this analysis. Ash is an important timber and wildlife habitat species, though it tends to not be a dominant species in North American forests. It rarely composes more than 25 percent of a stand's basal area (BA); it composes >50 percent of stand BA in only 8 percent of all ash-containing treelists in the analysis area.

Simulations were run for 35 years, (2008-2043) both with and without simulating EAB mortality,

using the Lake States (LS), Central States (CS), and Northeast (NE) variants of FVS (Version 1778), depending upon the geographic origin of each treelist. Simulations were run using 5-year cycles. No regeneration was simulated, except for possible stump-sprouting by “killed” ash. Stump sprouting routines were not modified. Structural Stage model output metrics were analyzed; differences between the with- and without-EAB simulations were compared. Structural Stage model parameters were left at their default values.

The EAB mortality was simulated via the SETPTHIN and THINPT keywords, which were parameterized to remove 40 percent of ash BA (>2 inches diameter at breast height, DBH) in the first 5-year cycle, and 80 percent of remaining ash (>2 inches DBH) over the second 5-year cycle, resulting in ~90 percent of the ash being “killed” over the first 10 years of the simulation—conservatively commensurate with rates published in Knight and others (2013). No subsequent ash mortality was simulated. The SETPTHIN and THINPT keywords were used because they allow specifying thinning by species groups (keyword SPGROUP); we defined our SPGROUP to be all ash species. Further, we used thinning, and not the FIXMORT keyword, to simulate mortality because FIXMORT precludes stump sprouting, a phenomenon we wanted to allow to happen in the simulation, as it is known to occur in EAB-“killed” ash.

We analyzed ecoregional scale trajectories of, and changes to, forest type, size class, and structural stage over 35 years. We present differences in forest type, structural stage, and size class between the

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with- and without-EAB simulations in simulation year (SY) 2043.

Approximately 10 percent of ash-containing treelists experienced a change in forest type (FT) in SY 2043 as a result of simulated EAB mortality. Some of the prevalent EAB-induced FT transitions are presented in table 1.

Size class changes were experienced in ~6 percent of treelists. Most transitions occurred between poletimber and sawtimber classes, approximately half in each direction.

Structural stage transitions were experienced in 6 percent of treelists. Transition types varied. The four predominant transitions included:

- From: stem exclusion To: understory re-initiation
- From old forest, single stratum To: old forest, multi-strata
- From: old forest, single-stratum To: stem exclusion
- From: stem exclusion To: old forest, single stratum

These transitions parallel the size-class changes observed and suggest perhaps that in many stands, ash exists predominantly in the understory, while in many other stands it exists predominantly in the overstory; hence, when ash is removed, the former stands would tend to increase in size class and structural stage, and in the latter the opposite would tend to occur.

Table 1—Predominant forest type transitions observed in SY 2043 between the No EAB and with EAB scenarios

“From” forest type (no-EAB, SY 2043)	“To” forest type (with EAB, SY 2043)
Black ash / American elm / red maple	Balsam fir
	Northern white-cedar
	Paper birch
	Red maple
Cherry / ash / yellow-poplar	Sugar maple / beech / yellow birch
	Sugar maple / beech / yellow birch
	Aspen
	Hard maple / basswood
Sugar maple / beech / yellow birch	Mixed upland hardwoods
	Red maple / oak
	Red maple
	White oak / red oak / hickory
	Sugarberry / hackberry / elm / green ash
Sugar maple / beech / yellow birch	Black ash / American elm / red maple
	Red maple / lowland
	Sugar maple / beech / yellow birch

EAB = emerald ash borer.

The analysis demonstrates the utility of FVS to address questions about the potential status of future forest landscapes vis-à-vis forest pests and other disturbances. Results presented demonstrate just a few of the types of compositional and structural attributes that could be analyzed via FVS. Additional analyses could include, for example, elucidating the resultant spatial mosaics of forest types and structures in context of their historic ranges of variability.

Our simulations removed 90 percent of ash BA over a 10-year period followed by 25 additional years of growth and only ‘background’ mortality. Knight and others (2013) found higher mortality rates in shorter periods of time; and EAB likely will continue to impose mortality on surviving ash into the future. Hence our simulations are conservative in estimating potential effects of EAB on ash-containing plant communities. Further, our simulations did not consider regeneration (e.g., seedling recruitment) in stands, except for stump sprouting. Incorporating such ecological phenomena would be important to consider in future analyses, especially those estimating potential changes many decades into the future. Accordingly, it would be fruitful if silviculturalists could build a library of stand-appropriate Regeneration Model (Ferguson and others 1991) keyword sets for use in FVS simulations. The creation of such a library could also incorporate potential climate-related processes on recruitment. Further, having available ‘wall-to-wall’ spatially imputed treelists would facilitate more robust and thorough spatial analyses of the potential effects of disturbances across landscapes.

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FSVeg Spatial Data Analyzer Projects

Using Landfire, FSveg Spatial Data Analyzer Nearest Neighbor, Forest Vegetation Simulator, and FlamMap to Compare Treatment Effects Across a Landscape

James Arciniega¹

Abstract—In order to simulate alternative future conditions, forest inventory data containing individual tree characteristics are required for programs that utilize Forest Vegetation Simulator (FVS). These ‘treelist’ data are not available nationally and are normally attained at a project scale. The Forest Service, U.S. Department of Agriculture, stores forest inventory data in the Field Sampled Vegetation (FSVeg) database. The spatial component of FSVeg (FSVeg Spatial) includes an application known as the FSVeg Spatial Data Analyzer (DA) which enables construction of a wall-to-wall vegetation dataset via a variety of nearest neighbor imputation methods (e.g., Most Similar Neighbor, Gradient Nearest Neighbor, Random Forest, etc.). Imputation is a process of “filling in” missing data with representative vegetation values using tested statistical methods from sampled data in or near the area of interest. Analysis of fire risk and fire hazard on a 10,000-acre project area was made possible in large part by imputation. The analysis was conducted using a combination of LANDFIRE and FSVeg Spatial DA imputed data where canopy characteristics were derived from FVS outputs. These data were combined into a landscape file and fire behavior metrics were derived from FlamMap outputs. Proposed Action effects to vegetation were modeled via FVS to derive post-activity fire behavior metrics from FlamMap outputs.

INTRODUCTION

Input for fire behavior programs such as FlamMap and FARSITE is readily accessible via the LANDFIRE database (www.landfire.gov). These data are useful for analysis of current conditions, but prove difficult to manipulate in a defensible and repeatable manner so as to reflect changes based on silvicultural manipulation. In order to simulate alternative future conditions, forest inventory data containing individual tree characteristics are required for programs that utilize Forest Vegetation Simulator (FVS) output files. These ‘treelist’ data are not available nationally and are normally attained at a project scale. Because of the lack of wall-to-wall data, many professionals use FVS to model treatments in a handful of stands and loosely extrapolate to an entire landscape. Using the methods outlined in this document, treatments can be simulated in every forested stand of a landscape.

Analysis of fire risk and fire hazard on the 10,000-acre Kiowa-San Cristobal Wildland-Urban Interface (KSC WUI) project area was made possible in large part with nearest neighbor imputation utilizing the Field Sampled Vegetation (FSVeg) Spatial

Data Analyzer (DA). The analysis was conducted using a combination of LANDFIRE and FSVeg data where canopy characteristics were derived from FVS outputs. These data were combined into a landscape file, and fire behavior metrics were derived from FlamMap (www.firelab.org) outputs for the No Action alternative. Proposed Action effects to vegetation were modeled via FVS to derive post-activity fire behavior metrics from FlamMap outputs, and a comparison of each alternative scenario was considered to determine recommendation of a preferred alternative.

IMPUTATION METHODS

The Forest Service stores forest inventory (Common Stand Exam; CSE) data within the FSVeg database and makes them spatially explicit by tying forest inventory data to vegetation polygons (i.e., stands) in the FSVeg Spatial application. The aforementioned DA is an analytical application that utilizes data from FSVeg and FSVeg Spatial to allow spatially explicit display and manipulation of forest inventory data. The DA enables construction of a wall-to-wall vegetation dataset, of particular utility to fire and fuels analysis, via `yaImpute`

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methods (Crookston and Finley 2008). The wall-to-wall vegetation dataset is produced in the DA after evaluation of a variety of nearest neighbor imputations (e.g., Most Similar Neighbor, Gradient Nearest Neighbor, Random Forest, etc.). Such extrapolation of data is necessary as forest inventory is impractical in every stand of a landscape-scale project, yet wall-to-wall tree data are required for thorough analysis of treatment effects and alternative future fire behavior. Imputation is a process of “filling in” missing data with plausible values. The DA uses a concept of an overarching nearest neighbor (NN) dataset (“Parent”) and a smaller project-scale NN dataset (“Child”). The Parent dataset is statistically evaluated based on many nearest neighbor imputations. If it is desirable to improve statistical correlation of imputed stands, the area can be stratified (“clustered”) using the DA to evaluate strategic placement of additional CSEs, whereby improving future iterations of imputation in the area of interest.

Parent NN Dataset

I created an imputed dataset for the project covering the Taos Valley Watershed Coalition area of interest (AOI, ~280,000 acres). Input for the AOI included LandSat8 imagery from September of 2013. In addition, Climate-FVS data and Digital Elevation Model rasters were utilized. FSveg stands were assessed for significant change between CSE collection date and LandSat8 imagery date. Several NN imputations were created and evaluated. The Most Similar Neighbor NN scenario was chosen as the best fit of selection variables. The project analysis was conducted from a second run on the Carson National Forest’s portion of the Taos Valley Watershed Coalition AOI (west slope Sangre De Cristo Mountains of northern New Mexico). After the first run it was determined that additional CSE data were needed. Stands were stratified (“clustered”) using the DA and improvement areas were identified for further data collection based on the clustering. The second run was the same as the first except for the addition of strategic CSEs from summer 2015. No significant disturbances occurred within the AOI in that 2-year gap between image date and latter CSE data collection. Clustering of stands can be particularly important when assessing large landscapes as imputation may be statistically acceptable as a whole even though particular stand types (e.g., forest cover types) may be poorly

imputed. This was generally apparent in Gambel oak (*Quercus gambelii*) and spruce/fir (*Picea* spp./*Abies* spp.) cover types where few stands had been previously inventoried, lacking data on the full range of variability within the project area. Identifying need for subsequent inventory may result in significant investment of time, effort, and planning as contracts and/or adjustments to program of work are needed to conduct additional forest inventory.

Child NN Project

A Child project was created from the Parent NN dataset. The Child NN project was created with alternative treatment scenarios so FVS calibrations could be instituted and so stand boundaries could be redrawn to match the project boundary (GIS overlay; a backdoor process used in lieu of a clean “clip” as at the time of project creation, the DA did not split stands to clip to a project boundary). This process resulted in some very minor over- and underage based on the DA’s methods. That is, some stands were not split if they exceeded the project boundary by less than two acres and other stands were completely omitted if only a small portion of the stand was within the project boundary. Strictly clipping to the project boundary would have resulted in hundreds, if not thousands, of tiny sub-acre slivers of stands along the project boundary as stand boundaries were not always coincident with the project boundary.

FVS CALIBRATION AND KEYWORDS

The Central Rockies (CR) Variant (Suppose v. 2.02- CR Variant v. 1675; Dixon 2002, Keyser and Dixon 2008) is the variant of FVS geographically appropriate for the Carson National Forest. Natural regeneration (seedling establishment) is not incorporated in CR. Growth and yield were adjusted by calibrating maximum density thresholds and regeneration. Wildfire conditions were also defined. Growth cycles are referred to in this section. They represent a time period in which accretion and mortality are calculated given various influences including management activities. The default growth cycle for CR is 10 years. All activities occur at the beginning of a growth cycle unless specified otherwise. Table 1 describes a timeline and keyword inputs used to simulate activities associated with the Proposed Action. See table 2 for a description

Table 1—FVS keywords used for alternative 2 projections in Kiowa-San Cristobal Wildland-Urban Interface

Alternative 2 – Proposed Action										
Year	Keyword	Description								
2016 PJ	MISTPREF	Species	PIPO	DMR 1	DMR 2	DMR 3	DMR 4	DMR 5	DMR 6	
		Removal Preference Weight		—	—	—	4,000	5,000	6,000	
	SPECPREF	All juniper, piñon pine and Gambel oak species removal preference = 999.								
	THINDBH	Diameter Class (inches)			Residual Trees Per Acre					
		0 – 3.0			65					
		3.1 – 6.0			22					
6.1 – 9.0			12							
		9.1 - 999			8					
HTGMULT	Gambel oak height multiplier of 0.5									
SPROUT	Gambel oak sprout multiplier of 0.25									
2016 PP & DMC	MISTPREF	Species	PIPO	DMR 1	DMR 2	DMR 3	DMR 4	DMR 5	DMR 6	
		Removal Preference Weight		1,000	2,000	3,000	4,000	5,000	6,000	
	SPECPREF	All juniper, piñon pine and Gambel oak species removal preference = 999. All fir species and Engelmann spruce removal preference = 80 Douglas-fir, blue spruce, bristlecone/limber/ponderosa/southwestern white pine removal preference = -80								
	YARDLOSS	Retain 10% of branchwood from felled trees in stand.								
MechPPDMC.kcp	Group selection up to 2 acres on 20% of stand. Low thin matrix to BA of 65 ft ² /acre.									
2016 Fuel- break	MISTPREF	Species	PIPO	DMR 1	DMR 2	DMR 3	DMR 4	DMR 5	DMR 6	
		Removal Preference Weight		—	—	—	4,000	5,000	6,000	
	SPECPREF	All juniper, piñon pine and Gambel oak species removal preference = 999. All fir species and Engelmann spruce removal preference = 80 Douglas-fir, blue spruce, bristlecone/limber/ponderosa/southwestern white pine removal preference = -80								
	YARDLOSS	Retain 100% of branchwood from felled trees in stand.								
THINBBA	Low thin 5-999" DBH to BA of 40 ft ² /acre.									
2016 Oak	THINCC	Mechanical thin all juniper, piñon, and Gambel oak species <5" DRC to 50% canopy cover.								
	HTGMULT	Gambel oak height multiplier of 0.5								
	SPROUT	Gambel oak sprout multiplier of 0.25								
2018	SIMFIRE	See table 4 for prescribed fire conditions. (70% of material jackpot burned in Fuelbreak & PJ units)								
2019	SIMFIRE	See table 4 for prescribed fire conditions. (80% of stand broadcast burned in all but Fuelbreak, PJ, and Oak units)								
2026	SIMFIRE	See table 3 for wildfire conditions. For smoke production utility only. Wildfire behavior and effects were not used for post-treatment metrics.								

of FVS keywords and calibrations used in the simulation.

FVS Keywords

SDIMAX—FVS determines maximum Stand Density Index (SDI) for a given site based on proportion of basal area per species represented within the stand. FVS methods were adjusted by using maximum SDI values as derived from national forest inventory data as per methods outlined by Long and Shaw (2005).

SDICALC—Stand Density Index was originally formulated by Reineke (1933) based on the quadratic mean diameter (QMD) of the stand. Stage (1968) presented a way to aggregate/disaggregate SDI to individual trees. Zeide (1983) recommended that the mean diameter used in calculating SDI should not be the QMD. Use of a method utilizing the summation of diameters other than QMD is recommended for stands without a normal (even-aged) diameter distribution (Curtis 2010, Long 1995, Shaw 2000); therefore, Zeide’s method was used in this analysis.

NATURAL—Consideration was given to natural regeneration rates in the project area and initiated at the rates described in table 2. Timber harvest activities were assumed to affect light concentrations in the understory, scarify soil, and reduce competition, so regeneration is induced 1 year after harvest activities.

HTGMULT/SPROUT—Number of sprouts and growth rates for Gambel oak were deemed excessive relative to observed rates in the area. Sprouts were induced after any activity and modified from base rates by a 0.25 multiplier. Height growth on Gambel oak was modified by a 0.5 multiplier as this species tends to remain shrub-like in stature, only occasionally attaining tree size in the project area.

YARDLOSS—Activity fuels are generated during timber harvest. By default, the Fire and Fuels Extension of FVS (FFE; Rebaun 2013) assumes all crown material associated with timber harvest is left in the stand. The YARDLOSS keyword was used to more accurately account for these additional fuels in the stand. It was assumed that 10 percent of crown material from felled trees is left in a stand if whole

tree yarding is specified (group selection and low thin within ponderosa pine (*Pinus ponderosa*) and dry mixed-conifer cover types).

MISTPREF—Preferentially remove trees by dwarf mistletoe rating. Higher values indicate preferential removal.

SPECREF—Adjust removal preferences by species group rather than individual species. Higher values indicate preferential removal, lower values indicate preferential retention.

SIMFIRE (Prescribed Fire)—Prescribed low intensity understory burning was used as a method of reducing shrub stature and continuity, reducing natural and activity fuel accumulations, and increasing canopy base height. This activity was modeled in year 2018 (pile/jackpot) and 2019 (understory) and assumed to cover 60-80 percent of the stand. Table 4 describes the assumed conditions during all prescribed fires. The moisture contents are derived from “moist” conditions as displayed in table 2.18 of the Fire and Fuels Extension to FVS (FFE-FVS) User Guide (Rebaun 2013). These conditions are in alignment with those common to local prescribed fire parameters.

SIMFIRE (Wildfire)—A wildfire was simulated in year 2026 in order to compare smoke emissions per alternative. This was simulated separately (i.e., an additional run with all of the same inputs and activities) and did not affect any of the FVS outputs used or quoted other than emissions. Potential climatic conditions were determined based on data from the Truchas Remote Automated Weather Station (RAWS; 290210) from the most recent 14-year period. Weather data collected by the RAWS were uploaded into Fire Family Plus version 4.0.2 (Bradshaw and McCormick 2000) in order to calculate percentile weather conditions. April 1 through July 31 weather data were considered over a period from 2002 to 2015 in order to calculate 97th percentile fuel and weather conditions. ‘Extreme’ fire conditions were not modeled as treatments are not assessed for effects during rare fire weather conditions.

FLAMMAP CALIBRATION

FlamMap v. 5.0.1.3 was utilized to simulate fire behavior based on inputs reflecting the No Action–

Table 2—FVS Calibration rationale-modeling of alternatives for Kiowa-San Cristobal Wildland-Urban Interface

Affected variable	FVS keyword	Calibration description
SDI	SDIMAX	634 – white fir (<i>Abies concolor</i>)
		423 – corkbark fir (<i>Abies lasiocarpa</i> var. <i>arizonica</i>)
		602 – subalpine fir (<i>Abies lasiocarpa</i>)
		411 – Rocky Mountain juniper (<i>Juniperus scopulorum</i>)
		620 – Engelmann spruce (<i>Picea engelmannii</i>)
		348 – two-needle piñon pine (<i>Pinus edulis</i>)
		409 – limber pine (<i>Pinus flexilis</i>)
		446 – ponderosa pine (<i>Pinus ponderosa</i> var. <i>scopulorum</i>)
		570 – Douglas-fir (<i>Pseudotsuga menziesii</i> var. <i>glauca</i>)
		562 – quaking aspen (<i>Populus tremuloides</i>)
	SDICALC	Summation method of calculating SDI for all trees ≥1" DBH
Regeneration	HTGMULT	Gambel oak height multiplier of 0.5 for all cycles
	SPROUT	Gambel oak sprout multiplier of 0.25 after any activity
	NATURAL	Timing W/in group selection opening
		Add X TPA Species Survival 5-yr. Height 450 PIPO 100% 2'
Removal Preference	MISTPREF	Within Timber Harvest Units (ponderosa pine and dry mixed-conifer)
		Species PIPO DMR 1 DMR 2 DMR 3 DMR4 DMR 5 DMR 6 Removal Preference Weight 1,000 2,000 3,000 4,000 5,000 6,000
	MISTPREF	Within Manual Thinning Units (fuelbreak, oak treatments, piñon/juniper)
		Species PIPO DMR 1 DMR 2 DMR 3 DMR4 DMR 5 DMR 6 Removal Preference Weight — — — 4,000 5,000 6,000
SPECPREF	Species Group* AJ AZ GO OJ CB ES AF WF BS BC DF LM PP SW PM PI RM UJ Removal Preference 999 80 -80	
Fuel Loading	YARDLOSS	During harvest activities, 10% of branchwood from harvested trees is left in the stand. 100% of it is on the ground.
RX Fire	SIMFIRE	Fire and fuel conditions assumed during prescribed fire implementation.
Wildfire	SIMFIRE	Fire and fuel conditions assumed during wildfire.

AJ—alligator juniper (*Juniperus deppeana*), AZ—Arizona piñon pine (*Pinus monophylla* var. *fallax*), GO—Gambel oak, OJ—oneseed juniper (*Juniperus monosperma*), PM—singleleaf piñon pine (*Pinus monophylla*), PI—two-needle piñon pine (*Pinus edulis*), RM—Rocky Mountain juniper, UJ—Utah juniper (*Juniperus osteosperma*), CB—corkbark fir, ES—Engelmann spruce, AF—subalpine fir, WF—white fir, BS—Colorado blue spruce (*Picea pungens*), BC—bristlecone pine (*Pinus aristata*), DF—Douglas-fir, LM—limber pine, PP—ponderosa pine, SW—southwestern white pine (*Pinus strobiformis*)

Table 3—97th percentile fuel moisture^a summary for Truchas RAWS (290210)

Fuel time-lag class	Moisture content (%)
1-hour	2
10-hour	3
100-hour	5
Live herbaceous	30
Live woody	60
Foliar	100

^a14-year average of April 1 to July 31, 2002-2015.

Current Conditions and the Action–Post-Treatment conditions. FlamMap is a fire behavior mapping and analysis program that computes potential fire behavior characteristics (spread rate, flame length, fireline intensity, etc.) across a user-defined landscape under constant weather and fuel moisture conditions. FlamMap fire behavior calculations are independent; that is, calculated fire intensity in one stand does not augment intensity in a neighboring stand. Potential fire behavior was modeled for the project area under warm, dry conditions. Weather and fuel moisture conditions were summarized from the Truchas RAWS and modeled under 97th percentile weather conditions. Fuel moistures used

in FlamMap fire behavior modeling are included in tables 3 and 4.

Various inputs are required and vary with desired outputs. All inputs that affect computational outputs used in the KSC WUI Fuels Specialist Report are displayed in table 5. Other inputs are required but are not discussed here as they only affect display and/or characteristics that were not used for the analysis. Creation of a landscape file is required for FlamMap fire behavior simulations. All required inputs to the landscape file were created and utilized as described in the Kiowa-San Cristobal WUI Fuels Specialist Report. They include: elevation, slope, aspect, fuel model, canopy cover, and the optional files for stand height, canopy base height, and canopy bulk density.

Fire Behavior

Fuel Moisture—Fuel moisture conditions were summarized from the Truchas RAWS, and 97th percentile fuel moisture conditions were calculated. Fuel moistures used for fire behavior modeling are displayed in table 3.

20-ft. Wind Speed—Twenty-foot wind speeds were summarized from the Truchas RAWS, and 97th percentile 20-foot wind speed conditions were calculated. The wind speed was adjusted slightly to

Table 4—Assumed variables for fire conditions in Kiowa-San Cristobal Wildland-Urban Interface

Variable	Description	Fire Scenario	
		Wildfire (97th %)	Prescribed (Moist/Fall)
1-hour	Moisture content (%) of dead surface fuel <0.25" diameter	2	5
10-hour	Moisture content (%) of dead surface fuel 0.25-1.00" diameter	3	6
100-hour	Moisture content (%) of dead surface fuel 1.01-3.00" diameter	5	8
1,000-hour	Moisture content (%) of dead surface fuel >3.00" diameter	8	15
*Duff	Moisture content (%) of duff (decomposed organic matter)	15	50
Live woody	Moisture content (%) of live woody material	60	90
Herb	Moisture content (%) of herbaceous material	30	30
20-foot wind	Wind speed (m.p.h.) at 20' above average vegetation height	23	6
Air temperature	Air temperature (°F) during fire	90	60

*Duff moistures are not reported by Fire Family Plus so they were derived from conditions described as 'very dry' and 'dry' in table 4.40 of the CR Variant User's Guide (Keyser and Dixon 2008).

Table 5—FlamMap inputs for Kiowa-San Cristobal Wildland-Urban Interface fire behavior metrics

Fire behavior inputs		Minimum travel time inputs	
Fuel moisture	(see table 3)	Random ignitions	2,500
20-foot wind speed	23 MPH	Wind direction	225°
Wind direction	Uphill	Resolution	60 m
Foliar moisture	100%	Simulation time	1,000 minutes
Crown fire calculation	Scott/Reinhardt	Spot probability	10%

be more representative of wildfire experienced in the area by increasing the value by about 15 percent.

Wind Direction—Fire behavior was influenced by an uphill wind for all cells in order to maximize potential rate of spread given constraints of all other inputs.

Foliar Moisture—One hundred percent live foliar moisture content was selected in order to approximate relatively dry season, but not yet dormant trees and shrubs.

Crown Fire Calculation Method—The Scott and Reinhardt (2001) crown fire calculation method is recommended if canopy bulk density values are generated from FFE-FVS (Stratton 2006) or LANDFIRE (www.landfire.gov). Depending on data type, Finney’s method under-predicts the likelihood a fire will transition to a crown fire with subsequent crown fire behavior activity compared to Scott and Reinhardt’s method (Scott 2006).

Conditional Burn Probability

Random Ignitions—Burn probabilities provide one method of evaluating the effectiveness of fuel treatments that removes the uncertainty of ignition sources. A large sample size of ignitions (say 1,000s or 10,000s) on the treated and pre-treatment landscape gives an indication of the overall effectiveness of the landscape pattern in retarding the growth of large fires. Multiple burn probability runs were executed in order to determine a suitable number of random ignitions. Two thousand five hundred random ignitions were reasonable for the landscape (~50,000 acres).

Wind Direction—Predominant winds on the Questa Ranger District are from the southwest. Wind direction was input as such (225°) for burn probability calculations as the intent was not to maximize potential rate of spread (as in the fire behavior runs), but to approximate probable wind direction.

Resolution—A 30-m resolution was utilized for fire behavior runs. During conditional burn probability simulation, a 60-m resolution was selected for its utility. Although increasing the resolution size may dilute some of the output values, it required a more reasonable timeframe for calculation. All outputs of simulated conditions for each alternative are compared for relative differences, so using the same methods for simulations of both alternatives avoids bias in outputs regardless of any dilution.

Simulation Time—This specifies duration (in minutes) of the fire growth calculations for the set of constant fuel moisture and wind conditions entered on the Inputs tab. One thousand minutes is representative of a fire burning for three days where the burning period is about five and one-half hours per day (3 days x 5.5 hours per day x 60 minutes per hour = 990 minutes).

Spot Probability—The default of 10 percent was selected in order to give consideration to the potential for fire spread to be affected by spotting, yet avoid an acceleration of fire behavior metrics that could be induced by a high spotting probability.

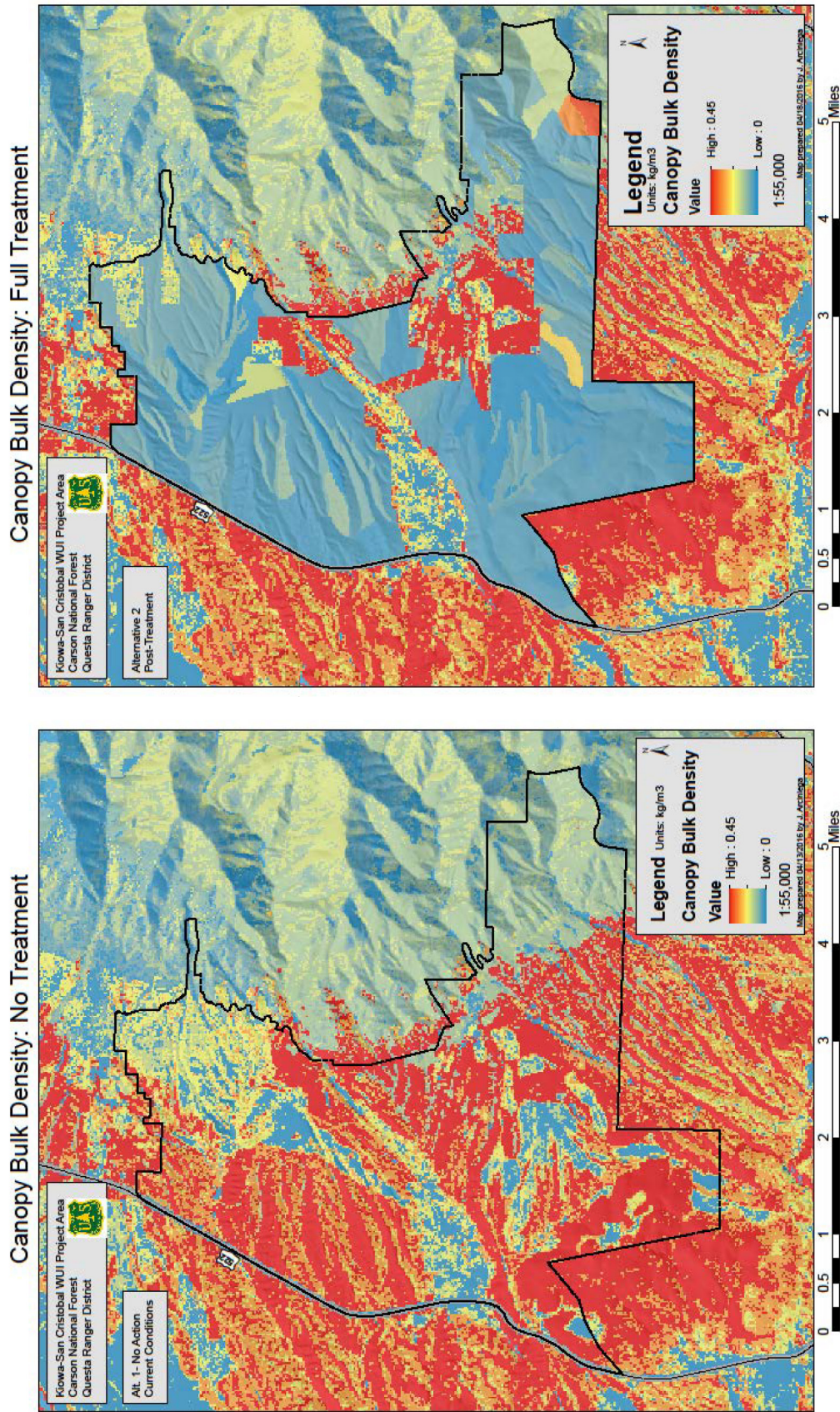


Figure 1—Canopy bulk density by alternative in Kiowa-San Cristobal Wildland-Urban Interface project area, Carson National Forest.

Table 6—Summary comparison of how the alternatives address the purpose and need

Purpose/need	Indicator/ measure	Desired relative condition	Measure	Alternative 1	Alternative 2
Wildfire resilience	2. Live Fuel Loading B. Canopy base height	High	CBH (feet)	Proportion (%)	Proportion (%)
			≤5	98	44
			5-15	<1	41
			15-25	<1	13
			>25	1	3
Wildfire resilience	2. Live Fuel Loading C. Canopy cover	Low	CC (%)	Proportion (%)	Proportion (%)
			≤20	14	57
			21-40	28	22
			41-60	57	21
			61-80	1	1
Wildfire resilience	2. Live fuel loading D. Canopy bulk density	Low	CBD (kg/m ³)	Proportion (%)	Proportion (%)
			<0.025	12	33
			0.025-0.050	<1	33
			0.050-0.075	2	10
			0.075-0.100	4	3
Wildfire resilience	3. Fire behavior A. Flame length	Short	FL (feet)	Proportion (%)	Proportion (%)
			≤5	33	43
			6-10	7	12
			11-20	5	19
			21-40	19	15
Wildfire resilience	3. Fire behavior C. Crown fire activity	Surface fire	CFA	Proportion (%)	Proportion (%)
			Unlikely fire	<1	<1
			Surface fire	40	46
			Passive crown fire	5	40
			Active crown fire	54	14

RESULTS/DISCUSSION

The methods outlined enabled assessment of spatially explicit stand variables. Such methods allow for a replicable wall-to-wall analysis using standard Forest Service programs, datasets, and protocols. This is particularly useful to fire and fuels analyses where data gaps compromise fire behavior assessment and loose extrapolation of treatment effects dilute relative differences in alternatives. See figure 1 for an example geographic representation of evaluated metrics by alternative. Table 6 provides summary output used for quantitative

comparison of alternatives and determination of the preferred alternative.

CONCLUSION

Use of FVS requires individual tree descriptive metrics to simulate changes in stand conditions from alternative silvicultural methods. Without imputed data, spatially explicit assessments cannot be made on a broad scale due largely to temporal and financial constraints requisite to obtaining field-collected individual tree data. The means of

producing wall-to-wall data is available through the FSveg Spatial Data Analyzer. While this process requires a high level of skills including experience in forestry, silviculture, fire suppression, fuels management, FSveg Spatial Data Analyzer, GIS, FVS, and FlamMap, many are normally available in interdisciplinary teams.

The FSveg Spatial Data Analyzer allowed for integrated data analysis to make a large landscape assessment possible within a reasonable time period. This included creating FVS-ready datasets from Common Stand Exam inventory, evaluation of vegetation conditions in areas lacking tree inventory by using nearest neighbor imputation, simulating treatments on a landscape with FVS, and production of FlamMap-related inputs for current and future conditions.

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Spatial Modeling of Timber Ecosystem Services: Linking the FVS Econ Extension and FSveg Spatial Data Analyzer to Map Stumpage Value

Christopher Haberland and Jonathan Marston¹

This paper documents an approach to quantify and map the potential economic benefits of timber harvest on a landscape by utilizing the FSveg Spatial Data Analyzer (DA) and the Economics Extension (ECON) of the Forest Vegetation Simulator (FVS). The DA application has been updated recently to integrate ECON, a module that dynamically interacts with FVS to output investment decision indices (Martin 2009). The linking of the DA and ECON allows users to map projected costs and benefits of tree harvest using FSveg stand exam data. To demonstrate this functionality, we estimate and assign average stumpage prices for different tree species on the Monongahela National Forest (MNF) and simulate the FVS “clearcut” management scenario across all stands. A clearcut scenario models the maximum timber market value for a stand, which represents the potential ecosystem service value, or economic benefit, of the timber for a given point in time. This spatially explicit output can assist forest planners and managers in comparing the value of ecosystem services provided by timber across stands of various compositions and at different spatial scales.

We used ECON’s HRVRVN keyword to assign timber prices for each of the 86 tree species categories in the Northeast variant (NE, version 1882) of FVS using average stumpage price data gleaned from regional surveys, as well as transactional evidence from 2016 timber sales on the MNF. This keyword file was globally applied to the entire MNF, 1,658,166 acres of Forest Service land. To estimate sawtimber value, prices are only applied to trees greater than 9 inches diameter at breast height (4.5 feet above groundline, dbh) for

softwood species and 11 inches dbh for hardwood species. Stumpage price data was collected from the following sources:

1. Quarterly stumpage prices from Appalachia Hardwood Center’s timber market report (Appalachian Hardwood Center 2013-2016) for the MNF region for the years 2013-2016 (“AHC” in table 1).
2. Quarterly stumpage prices from the PennState Extension timber market report (PennState 2013-2016) for southwestern region of Pennsylvania for the years 2013-2016 (“Penn” in table 1).
3. Semiannual stumpage prices from the Ohio timber price reports (Ohio State 2013-2016) for the years 2013-2016 (“Ohio” in table 1).
4. Transaction evidence data (U.S. Department of Agriculture 2016) available to Forest Service (FS) timber contracting officers for the year 2016 (“FS TE” in table 1). (To better approximate a species’ market value, this data was multiplied by a scaling factor of 3.02, the median ratio of prices between species common to price groups in both the Transaction Evidence database and the Appalachia Hardwood Center’s timber market report data).

Lumber from each species is assumed to have at least some value, with a floor price of \$43/MBF applied to less commercially important species groups that together comprise roughly 2 percent of the total modeled volume across all stands. This price is derived from the 2013-2016 average price of lumber in the “Other” category from the

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Table 1—Price and volume estimates by species for all modeled stands on the Monongahela National Forest for 2017

Common name in NE variant	Scientific name	Total merchantable volume Sawtimber, thousand board feet (MBF)	Percentage of total estimated volume % of total across all stands	Stumpage price 2016 dollars per MBF	Price data source	Price species group	Percentage of total estimated value % of total across all stands
Black cherry	<i>Prunus serotina</i> Ehrh.	1,244,181	14.69845	460.26	AHC	Black cherry	29.935038
Northern red oak	<i>Quercus rubra</i> L.	1,244,234	14.69908	307.24	AHC	Red oak	19.979231
Sugar maple	<i>Acer saccharum</i> Marsh.	530,992	6.27301	398.89	AHC	Hard maple	11.081532
Red maple	<i>Acer rubrum</i> L.	612,150	7.23180	250.50	AHC	Soft maple	8.036518
Yellow-poplar	<i>Liriodendron tulipifera</i> L.	704,936	8.32794	189.38	AHC	Yellow-poplar	6.968746
Chestnut oak	<i>Quercus prinus</i> L.	683,088	8.06984	140.43	FS TE	Chestnut oak	5.002106
White oak	<i>Quercus alba</i> L.	225,639	2.66565	251.74	AHC	White oak	2.974073
Red spruce	<i>Picea rubens</i> Sarg.	418,299	4.94169	98.25	Penn	Hemlock	2.144162
Eastern hemlock	<i>Tsuga canadensis</i> (L.) Carrière	342,411	4.04516	98.25	Penn	Hemlock	1.755157
American beech	<i>Fagus grandifolia</i> Ehrh.	567,993	6.71014	58.84	FS TE	American beech	1.752820
Cucumber tree	<i>Magnolia acuminata</i> (L.) L.	167,856	1.98302	189.38	AHC	Yellow-poplar	1.659312
Black oak	<i>Quercus velutina</i> Lam.	96,332	1.13804	282.67	FS TE	Black oak	1.425998
Scarlet oak	<i>Quercus coccinea</i> Münchh.	130,284	1.53915	173.22	FS TE	Scarlet oak	1.178894
White ash	<i>Fraxinus americana</i> L.	153,628	1.81493	141.17	AHC	Ash	1.133000
American basswood	<i>Tilia americana</i> L.	325,107	3.84073	65.93	FS TE	Basswood	1.122346
Yellow birch	<i>Betula alleghaniensis</i> Britton	237,641	2.80744	64.55	FS TE	Yellow birch	0.807935
Eastern white pine	<i>Pinus strobus</i> L.	135,173	1.59690	108.58	Penn	White pine	0.770640
Sweet birch	<i>Betula lenta</i> L.	120,347	1.42175	50.86	FS TE	Sweet birch	0.320987
Shagbark hickory	<i>Carya ovata</i> (Mill.) K. Koch	65,577	0.77471	93.21	AHC	Hickory	0.318994
Pitch pine	<i>Pinus rigida</i> Mill.	80,907	0.95581	73.91	Ohio	Pine	0.313108
Hickory species	<i>Carya</i> Nutt.	52,262	0.61741	93.21	AHC	Hickory	0.254206
Red pine	<i>Pinus resinosa</i> Aiton	55,164	0.65169	73.91	Ohio	Pine	0.213516
Magnolia species	<i>Magnolia</i> L.	50,646	0.59832	58.83	FS TE	Magnolia spp.	0.156323
Norway spruce	<i>Picea abies</i> (L.) Karst.	29,780	0.35181	98.25	Penn	Hemlock	0.152636
Black locust	<i>Robinia pseudoacacia</i> L.	63,800	0.75372	43.03	AHC	Other	0.143461
Blackgum	<i>Nyssa sylvatica</i> Marsh.	31,354	0.37041	43.03	AHC	Other	0.070557
Virginia pine	<i>Pinus virginiana</i> Mill.	17,351	0.20498	73.91	Ohio	Pine	0.067171
Mockernut hickory	<i>Carya alba</i> (L.) Nutt.	13,706	0.16192	93.21	AHC	Hickory	0.066662
Black walnut	<i>Juglans nigra</i> L.	2,430	0.02870	312.36	AHC	Walnut	0.039646
Bigtooth aspen	<i>Populus grandidentata</i> Michx.	16,308	0.19266	43.03	AHC	Other	0.036671

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Table 1 (Continued)—Price and volume estimates by species for all modeled stands on the Monongahela National Forest for 2017

Common name in NE variant	Scientific name	Total merchantable volume Sawtimber, thousand board feet (MBF)	Percentage of total estimated volume % of total across all stands	Stumpage price 2016 dollars per MBF	Price data source	Price species group	Percentage of total estimated value % of total across all stands
Other hardwoods		10,894	0.12870	43.03	AHC	Other	0.024464
River birch	<i>Betula nigra</i> L.	4,507	0.05325	43.03	AHC	Other	0.010156
Sycamore	<i>Platanus occidentalis</i> L.	4,052	0.04787	43.03	AHC	Other	0.009125
Buckeye species	<i>Aesculus</i> L.	3,191	0.03769	43.03	AHC	Other	0.007180
Striped maple	<i>Acer pensylvanicum</i> L.	530	0.00626	250.50	AHC	Soft maple	0.006951
Table-mountain pine	<i>Pinus pungens</i> Lamb.	2,745	0.03242	43.03	AHC	Other	0.006179
American elm	<i>Ulmus americana</i> L.	2,607	0.03079	43.03	AHC	Other	0.005860
Sassafras	<i>Sassafras albidum</i> (Nutt.) Nees	2,563	0.03028	43.03	AHC	Other	0.005764
Silver maple	<i>Acer saccharinum</i> L.	413	0.00488	250.50	AHC	Soft maple	0.005424
Quaking aspen	<i>Populus tremuloides</i> Michx.	2,409	0.02846	43.03	AHC	Other	0.005414
Other elm species	<i>Ulmus</i> L.	2,299	0.02716	43.03	AHC	Other	0.005162
Non-commercial hardwoods		1,390	0.01643	43.03	AHC	Other	0.003127
Pignut hickory	<i>Carya glabra</i> (Mill.) Sweet	617	0.00729	93.21	AHC	Hickory	0.002992
Slippery elm	<i>Ulmus rubra</i> Muhl.	1,155	0.01364	43.03	AHC	Other	0.002605
Sourwood	<i>Oxydendrum arboreum</i> (L.) DC.	896	0.01058	43.03	AHC	Other	0.002030
Pin cherry	<i>Prunus pensylvanica</i> L. f.	779	0.00920	43.03	AHC	Other	0.001776
Butternut	<i>Juglans cinerea</i> L.	680	0.00804	43.03	AHC	Other	0.001524
Serviceberry	<i>Amelanchier</i> Medik.	518	0.00612	43.03	AHC	Other	0.001161
Pin oak	<i>Quercus palustris</i> Münchh.	133	0.00158	155.00	AHC	Mixed oak	0.001079
Southern red oak	<i>Quercus falcata</i> Michx.	65	0.00077	307.24	AHC	Red oak	0.001044
Cherrybark oak	<i>Quercus pagoda</i> Raf.	62	0.00074	307.24	AHC	Red oak	0.000998
Chinkapin oak	<i>Quercus muehlenbergii</i> Engelm.	70	0.00083	251.74	AHC	White oak	0.000921
Shortleaf pine	<i>Pinus echinata</i> Mill.	199	0.00235	73.91	Ohio	Pine	0.000768
Eastern cottonwood	<i>Populus deltoides</i> Bartram ex Marsh.	274	0.00324	43.03	AHC	Other	0.000619
Water tupelo	<i>Nyssa aquatica</i> L.	257	0.00304	43.03	AHC	Other	0.000578
Other oak species	<i>Quercus</i> L.	68	0.00081	155.00	AHC	Mixed oak	0.000555
Black maple	<i>Acer nigrum</i> Michx. f.	26	0.00031	398.89	AHC	Hard maple	0.000542
Eastern redcedar	<i>Juniperus virginiana</i> L.	213	0.00251	43.03	AHC	Other	0.000477

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Table 1 (Continued)—Price and volume estimates by species for all modeled stands on the Monongahela National Forest for 2017

Common name in NE variant	Scientific name	Total merchantable volume Sawtimber, thousand board feet (MBF)	Percentage of total estimated volume % of total across all stands	Stumpage price 2016 dollars per MBF	Price data source	Price species group	Percentage of total estimated value % of total across all stands
Boxelder	<i>Acer negundo</i> L.	213	0.00251	43.03	AHC	Other	0.000471
Bur oak	<i>Quercus macrocarpa</i> Michx.	55	0.00065	155.00	AHC	Mixed Oak	0.000446
Sweetgum	<i>Liquidambar styraciflua</i> L.	152	0.00180	43.03	AHC	Other	0.000342
Common persimmon	<i>Diospyros virginiana</i> L.	147	0.00174	43.03	AHC	Other	0.000335
Hackberry	<i>Celtis occidentalis</i> L.	140	0.00165	43.03	AHC	Other	0.000320
Balsam fir	<i>Abies balsamea</i> (L.) Mill.	120	0.00142	43.03	AHC	Other	0.000271
Swamp white oak	<i>Quercus bicolor</i> Willd.	32	0.00038	155.00	AHC	Mixed oak	0.000265
Apple species	<i>Malus</i> Mill.	118	0.00140	43.03	AHC	Other	0.000258
Eastern hophornbeam	<i>Ostrya virginiana</i> (Mill.) K. Koch	101	0.00119	43.03	AHC	Other	0.000238
Scotch pine	<i>Pinus sylvestris</i> L.	48	0.00057	73.91	Ohio	Pine	0.000188
Yellow buckeye	<i>Aesculus flava</i> Aiton	70	0.00083	43.03	AHC	Other	0.000157
Hawthorn species	<i>Crataegus</i> L.	64	0.00076	43.03	AHC	Other	0.000145
American holly	<i>Ilex opaca</i> Aiton	64	0.00076	43.03	AHC	Other	0.000145
Northern white-cedar	<i>Thuja occidentalis</i> L.	40	0.00047	43.03	AHC	Other	0.000089
Balsam poplar	<i>Populus balsamifera</i> L.	38	0.00045	43.03	AHC	Other	0.000087
Ash species	<i>Fraxinus</i> L.	9	0.00010	141.17	AHC	Ash	0.000063
Green ash	<i>Fraxinus pennsylvanica</i> Marsh.	7	0.00008	141.17	AHC	Ash	0.000049
Black ash	<i>Fraxinus nigra</i> Marsh.	5	0.00006	141.17	AHC	Ash	0.000040
American hornbeam	<i>Carpinus caroliniana</i> Walter	18	0.00021	43.03	AHC	Other	0.000039
Loblolly pine	<i>Pinus taeda</i> L.	9	0.00010	73.91	Ohio	Pine	0.000033
Flowering dogwood	<i>Cornus florida</i> L.	14	0.00016	43.03	AHC	Other	0.000032
Tamarack	<i>Larix laricina</i> (Du Roi) K. Koch	13	0.00016	43.03	AHC	Other	0.000031
White basswood	<i>Tilia americana</i> L. var. <i>heterophylla</i> (Vent.) Louden	9	0.00010	65.93	FS TE	Basswood	0.000030
Other cedar species	<i>Juniperus</i> L.	13	0.00015	43.03	AHC	Other	0.000030
Post oak	<i>Quercus stellata</i> Wangenh.	3	0.00004	155.00	AHC	Mixed Oak	0.000028
Black willow	<i>Salix nigra</i> Marsh.	8	0.00010	43.03	AHC	Other	0.000021
White spruce	<i>Picea glauca</i> (Moench) Voss	4	0.00005	98.25	Penn	Hemlock	0.000020
Paper birch	<i>Betula papyrifera</i> Marsh.	2	0.00003	43.03	AHC	Other	0.000005

Appalachia Hardwood Center’s timber market report series. The valuation of lumber in this category reflects the fact that several species not commercially milled in large quantities are sold in small scales. We set the SDICALC keyword to the Zeide SDI calculation method, and the PRETEND and THINDBH keywords to model a hypothetical clearcut scenario to calculate potentially harvestable volume for the year 2017. These keywords were loaded into the DA and each stand was “grown” in NE to the year 2017 using data from its most recent FSveg exam. Stands without any record of FSveg examination were neither simulated nor included in forest summary statistics.

Figure 1 shows the potential revenue resulting from clearcutting each stand, measured in 2016 dollars per acre. This output does not account for any costs of harvest, as the purpose is to determine

the maximum ecosystem service value of timber harvest. As expected, sawtimber stumpage value is simulated as highest in the central region of the MNF where there is a comparatively greater concentration of mature hardwood stands comprised of oak (*Quercus* spp.) and maple (*Acer* spp.). The northeastern and southeastern regions of the forest, as well as the area surrounding Cheat Mountain (38°23’37”N 79°59’02”W), are modeled as the least valuable. These regions contain higher concentrations of softwood species according to the most recent FSveg surveys. The median stumpage value from the population of all modeled stands was \$2,507 per acre, and the mean value was \$3,042 per acre. These values are comparable to per acre stumpage estimates of Appalachia-region stands cited in previous studies (Burger and others 2012, Moss and Heitzman 2013) when adjusted for inflation.

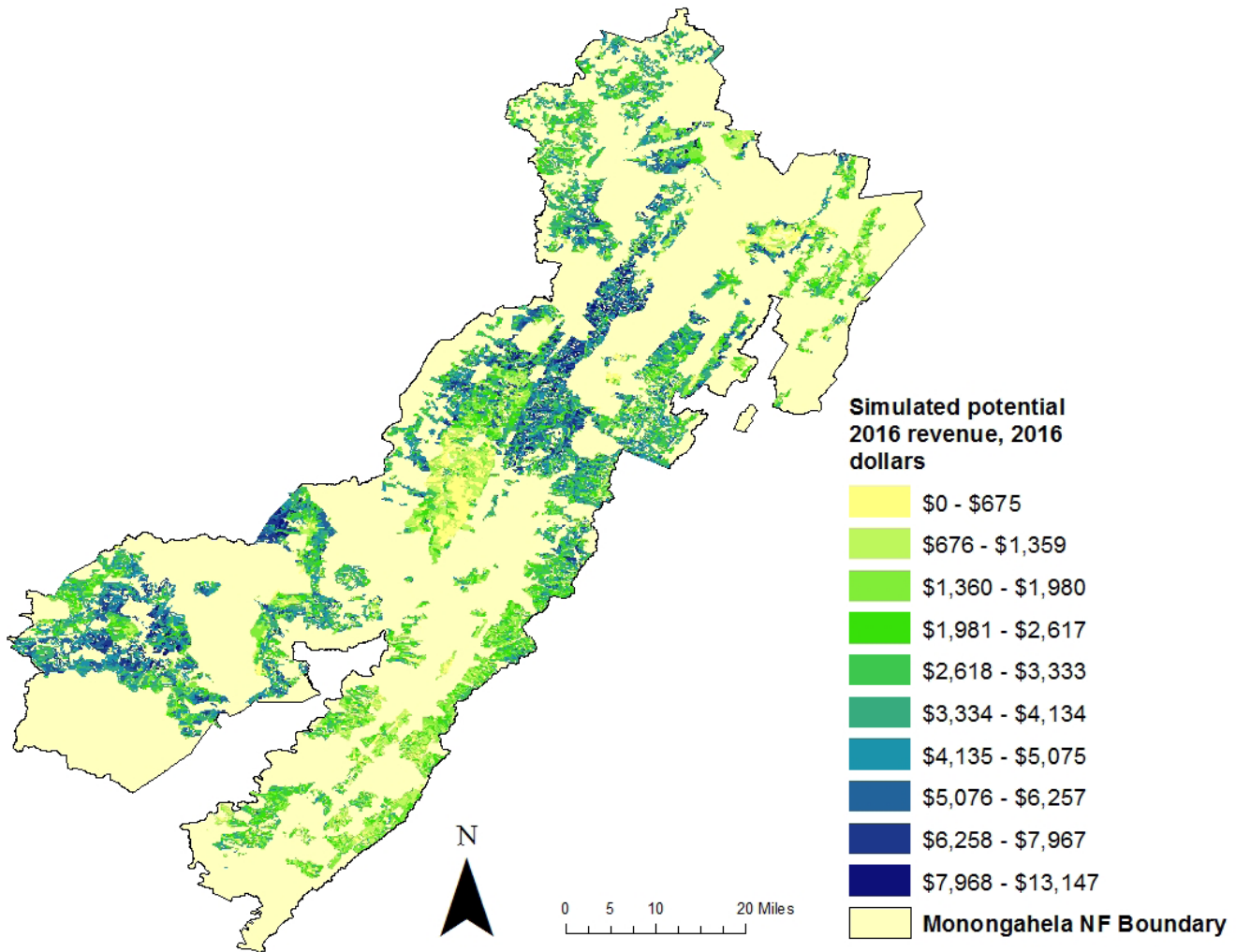


Figure 1—Simulated 2017 stumpage value across all surveyed stands in the Monongahela National Forest.

The generalizability of the results is limited by necessary design simplifications. We did not account for any logging or transportation costs, as we sought only to quantify and map the potential economic *benefits* of timber harvest. We also did not consider tree or sawlog grade differentiation among different stands, which would affect log prices. Although it is possible to estimate tree grade based on site index and other stand characteristics (Miller and others 2008, Yaussey 1993), no work has yet been conducted to link grade estimates into ECON outputs. Another limitation is that the results can only be interpreted in the context of a static timber market. The revenue outputs do not consider the local timber market's response to any sale on the market. Therefore, the summation of the model output across all stands does not reflect an estimate of the total value of timber ecosystem services in all stands for a single point in time. Rather, the results are useful for estimating spatial variation of timber ecosystem services between stands for the year 2017. We did not simulate future values or model alternative management practices and uncertainty about future timber prices in the MNF region due to time constraints. However, it is entirely possible to use market price projections and apply various future management scenarios with ECON and DA to simulate the effects on timber value.

These results demonstrate the usefulness of the DA and ECON for characterizing ecosystem services from harvested timber. Natural resource managers can compare the spatially explicit outputs generated by this process with other spatial data to evaluate the implications of different alternatives and scenarios. This method allows analysts to identify tradeoffs within a landscape by visualizing the spatial coincidence of ecosystem services that should be considered when making investments or siting projects.

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Economics Projects

Economic Returns of White Spruce Plantation Thinning Scenarios Using Forest Vegetation Simulator (FVS)

Curtis L. VanderSchaaf, Gordon Holley, and Joshua Adams¹

Abstract—Out of the approximate 88,000 acres of Minnesota white spruce plantations [*Picea glauca* (Moench) Voss], one-fifth of the acreage is managed by the Minnesota Department of Natural Resources. Many of these plantations are at or near the time for a potential first thinning, and some for a potential second thinning. Hence, the objective was to use the Forest Vegetation Simulator (FVS) to determine the optimal number of thinnings, residual stand density following thinnings, and final harvest rotation age to maximize economic returns. For simplicity, it was assumed that all harvested timber was white spruce. Four different thinning treatment scenarios and an unthinned scenario were examined. Thinning scenarios differed as to the timing of thinnings based on standing basal area per acre and the residual basal area per acre following the thinning. Timings of final harvests were modeled based on maximizing financial returns for the differing thinning times and intensity scenarios. When using stumpage revenues of \$12.64, \$20.58, and \$40.46 per cord for pulp only, pulp and bolt, and sawlog, respectively, the optimum financial regime on the lower site index site (SI of 59, feet-base age 50) was thin from 150 square feet to 120 square feet of basal area per acre and to conduct a final harvest at age 60. In comparison, on the higher site index site (SI of 70) it was optimum not to thin and conduct a final harvest at age 50. A more operationally feasible regime of thinning at 150 square feet to 90 square feet of basal area per acre was nearly as financially optimum on both site qualities.

INTRODUCTION

White spruce [*Picea glauca* (Moench) Voss] plantations on Minnesota Department of Natural Resources (DNR) lands are an essential source of fiber for pulp and paper mills, and the timber is also highly desired by sawmills (VanderSchaaf and others 2016). According to the DNR's Forest Inventory Module (FIM-03/08/2012), plantations exist on 2,292 stands covering 36,298 acres. Most stands are in mid-rotation age classes of 25 to 45 years (fig. 1) and are mainly in northern Minnesota. The DNR owns approximately 22 percent of the State's spruce plantation acres.

Pure stands of naturally-regenerated white spruce are not common. However, due to the importance of white spruce to the production of paper, many plantations were established in the 1960s and henceforth. Despite white spruce's importance, there has been minimal assessment of the growth and yield associated with various thinning regimes. Compared to other cover types, across all ownerships, there is a high percentage

of acres where management objectives will soon require a thinning (e.g., age class 25 and 35). With recent State budget issues, DNR management is under greater scrutiny by the public, particularly on School Trust lands whose generated revenues are used to support schools. Additionally, current concerns about pathogens (e.g., D'Amato and others 2011, Russell and others 2015), particularly eastern spruce budworm (*Choristoneura fumiferana* Clemens), make quantifying future growth rates and economic returns, particularly in response to first and second thinnings, important at the current time.

For white spruce, the DNR generally recommends rotation ages from around 60 to 90 years, with a first thinning occurring on site indexes (base age 50) of 60 and greater at around age 25 or 30 to a residual basal area between 110 to 120 square feet per acre. No more than two thinnings is recommended, in part because this species has a shallow root system and is easily damaged by heavy equipment. The Wisconsin DNR recommends an initial thinning at a target basal area of 160 to 90 square feet per acre

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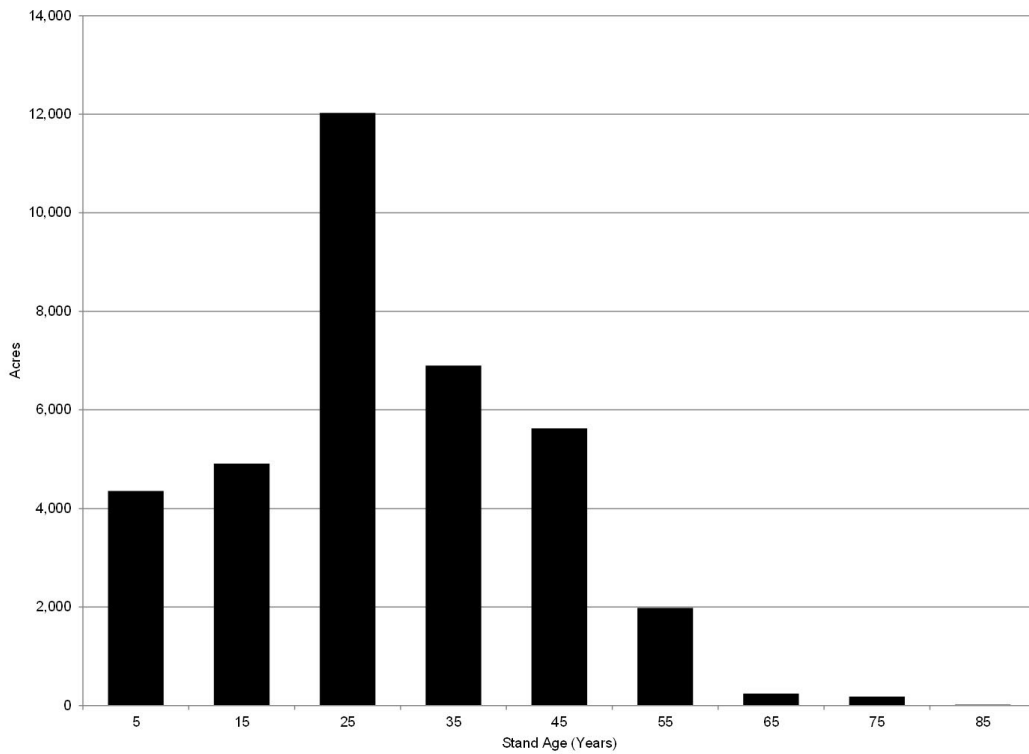


Figure 1—Existing harvestable (excludes old growth designated stands and other stands designated as non-harvestable) white spruce plantation cover type (Stand Origin of 2) age-class distribution (FIM 03/08/2012). Total number of stands is 2,292, and the total number of acres is 36,298. An age class of 35 corresponds to stands aged 31 to 40 year.

while trees are in the pole stage, but residual basal area should be 120 square feet for sawtimber size-dominated stands.

The objectives of this study were to use the Forest Vegetation Simulator (FVS) to help determine the optimal number of thinnings, the residual stand density following thinnings, and the final harvest age to maximize economic returns of white spruce plantations in northern Minnesota.

METHODS

The Lake States (LS, version 4862) variant of FVS covers Minnesota (Dixon and Keyser 2008). The Bareground option was used to generate a plantation of 450 trees per acre with a site index of 59 feet (base age 50) which represents the weighted acreage average site index of DNR plantations based on FIM. Initial runs showed negative financial returns, thus site index 70 was also examined to see if positive financial returns will be seen on higher quality sites (site index of 70 and greater occurs on 176 stands and 3,373 acres). Survival at age 10

was 100 percent. Discounted regeneration costs were \$455 per acre, including \$150 per acre for site preparation, \$225 per acre for 450 seedlings and planting, and \$80 per acre for a year one release treatment.

Stump height was 1 foot, minimum merchantable pulpwood class diameter at breast height (4.5 feet above groundline, DBH) was 5.0 inches, and upper stem diameter inside bark (DIB) was 4.0 inches. In addition, the bolt and pulp class volume was specified as minimum DBH of 8.0 inches with a maximum DBH of 11 inches, and upper stem DIB was 4 inches. Sawlog class volume was defined as trees with a minimum DBH of 12 inches and greater to a 7.6-inch top DIB. Pulpwood class volume was specified as all merchantable volume obtained from trees with diameters smaller than 8.0 inches and all volume on sawlog-sized trees from the 7.6-inch top to the 4.0-inch top.

Stumpage prices were \$12.64, \$20.58, and \$40.46 per cord, respectively, for pulpwood, bolt and pulp, and sawlog sized trees. These values represent the

DNR white spruce stumpage prices for the 2012 calendar year. Appraisal/marketing costs of \$12 and \$6 per cord were assumed during thinnings and clearcuts, respectively. A 3-percent interest rate [Minnesota Management and Budget (MMB)] was used to produce Soil Expectation Value (SEV). The default white spruce FVS maximum basal area of 190 square feet per acre and its associated maximum Stand Density Index (SDI) of 410 was used in the simulations. At least 10 years was required following a thinning before a final harvest could be conducted.

A total of four thinning treatment scenarios and a no thinning scenario were examined (table 1). All thinnings were assumed to be from below (i.e., the removal of trees from the lower crown classes favoring those in the upper crown classes). When a stand trajectory reached the target basal area, it was thinned in that year.

RESULTS

The greatest standing volume occurred in the unthinned treatment, with the SI 70 producing more volume than SI 59 (fig. 2). Estimated basal areas and volumes per acre by thinning treatments show logical and typical progressions throughout time.

To verify unthinned FVS projections, merchantable yield tables (Bell and others 1990) were examined of unthinned plantations in northwestern Ontario. Generally, FVS overpredicted volume, but the verification shows predicted yields are comparable.

In a yield table developed for Ontario, on sites planted with 436 seedlings per acre, Rauscher (1984) reported 56 cords per acre for a site index 70 (base age 50) site at age 50. This is similar to what FVS predicts.

Figure 3 shows cumulative volumes (total of the harvested and standing). The unthinned trajectory does not include the capture of mortality and is thus standing volume at a particular point in time. Based on assumed regeneration methods, merchantability standards, etc., and growth and yield projections from FVS, thinning increases merchantable stand volume production over time. At ages near DNR rotation ages (e.g., 50 to 60), optimum thinning regimes vary slightly by site quality. On lower site qualities (e.g., SI 59 feet), thinning to a residual basal area of 90 square feet per acre seems to maximize merchantable stand volume production, but results are basically the same for 150 and 120 target basal areas. However, for better sites (e.g., SI 70 feet), carrying relatively high densities appears best (150 square feet of basal area per acre), with the optimum residual basal area around 90 square feet of basal area per acre; this is quite similar to Wisconsin's DNR (Wisconsin DNR 2013) recommendations (target basal area of 160 square feet with 90 residual). The thinning from 120 to 60 square feet of basal area thinning treatment did not result in enough utilization of the site, but it does produce relatively larger trees (fig. 4).

By age 40, most trees in all treatments and on both sites are bolt size which demands a relatively

Table 1—Description of the four thinning treatment scenarios and the unthinned treatment scenario

Treatment	Description	Representation
1	Unthinned	Unthinned
2	Once stand BA reaches 120 square feet per acre thin back to 60 square feet.	120_60
3	Once stand BA reaches 120 square feet per acre thin back to 90 square feet (probably most common operationally).	120_90
4	Once stand BA reaches 150 square feet per acre thin back to 120 square feet.	150_120
5	Once stand BA reaches 150 square feet per acre thin back to 90 square feet.	150_90

All thinnings were assumed to be from below. BA is basal area per acre.

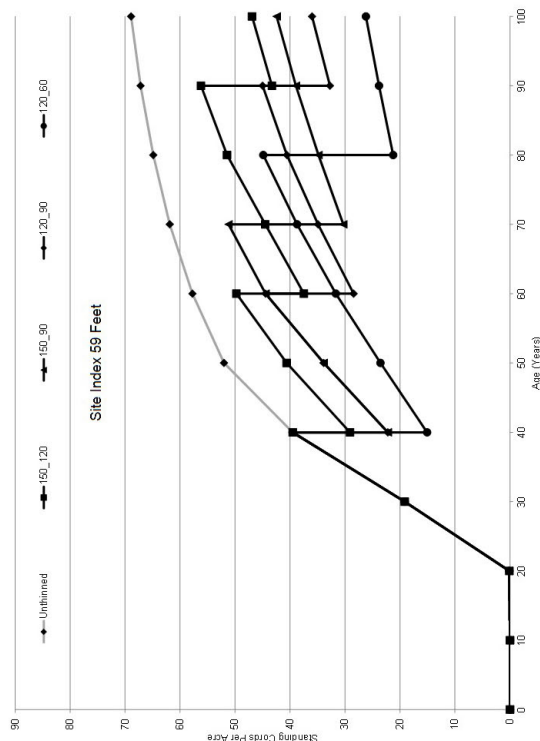
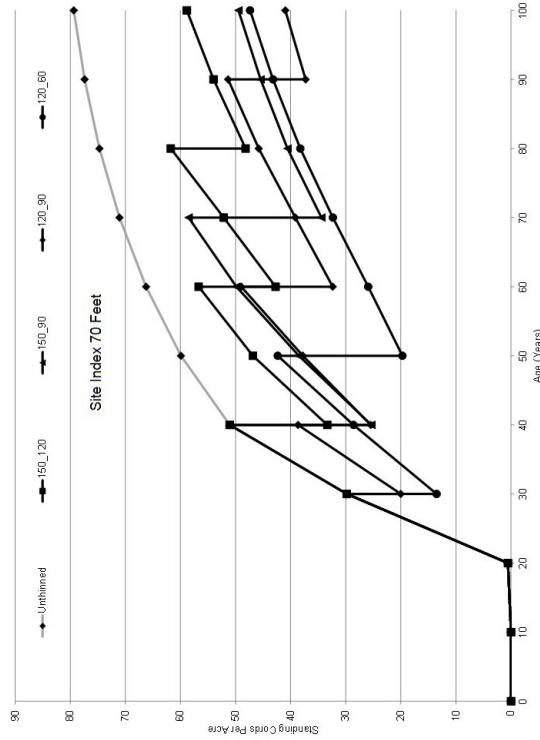


Figure 2—Merchantable cords per acre by thinning treatment. Merchantability is defined as all volume above a 1-foot stump to a 4-inch top DBH on all trees with DBH's of 5 inches or greater. Thinning treatments are defined in table 1.

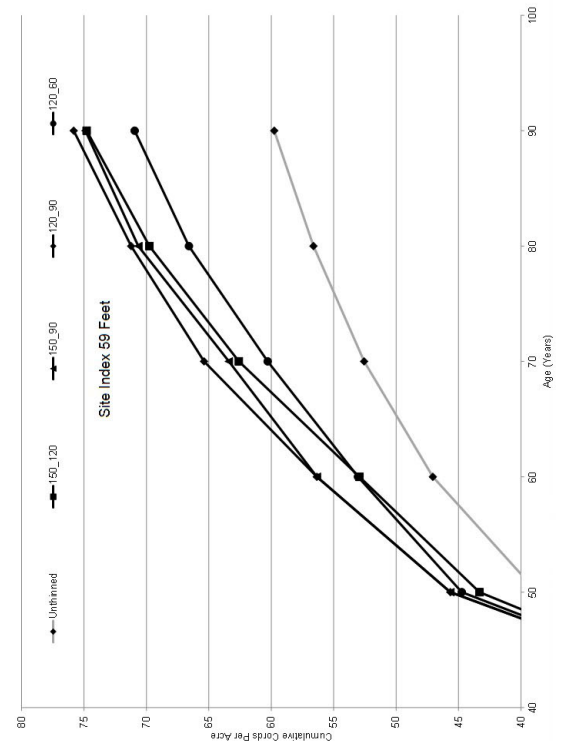
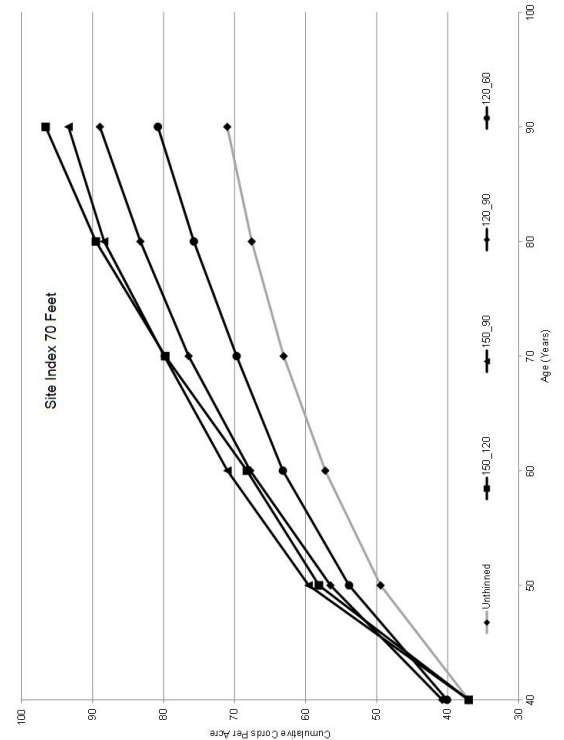


Figure 3—Cumulative merchantable cords by treatment. Thinning treatments are defined in table 1.

greater stumpage value than exclusively pulpwood (\$12.64 per cord versus \$20.58 per cord). In terms of quadratic mean diameter (Dq, inches), results from FVS do show meaningful differences among thinning treatments, but thinning vastly increases average tree diameter.

Figure 5 shows that despite the ability of the high intensity thinning (120 to 60 square feet of residual basal area per acre) to produce larger trees, the site was not utilized to its capacity and given current markets, produced the least optimum economic return on both sites. For the low quality site, the optimum regime was 150_120 at age 60 (\$-211.35); if stands remain unthinned, a final rotation at age 50 appears best (\$-216.75).

For higher quality sites, given current markets, it appears best not to thin (optimum rotation age of 50 years with a return of \$-80.59). The best thinning regime was the 150_120 treatment (\$-106.71 at age 50). Economic results differ from the cumulative volume results, the product breakdown given current prices produces these differences.

DISCUSSION

Financial Assessment

In all cases, planting white spruce will not return a positive investment at a 3-percent interest rate. The internal rate of return (IRR) for the unthinned treatment on the high quality site is 2.70 percent (rotation age of 50 years), the IRR on the lower quality site for the 150_120 treatment is 2.15 percent (rotation age of 60 years). Hence, based on this analysis, if plantations are to be established purely for the production of white spruce volume ignoring financial losses, these should only be established on high quality sites (e.g., site index of 70 and greater) which produce greater yields. Beyond simply stand-level financial concerns though, the DNR has a responsibility to provide fiber for industry and to produce ecological diversity across the landscape. Although the DNR may lose revenues establishing these plantations, these forests provide valuable fiber to industry helping to maintain a steady wood supply and a viable forest industry and provide employment and tax revenues.

To show the sensitivity of the optimal harvest regime to stumpage values, prices from the DNR 2011 fiscal year for white spruce were used: \$15.77, \$24.22, and \$32.12 per cord for pulpwood, bolt and pulpwood, and sawtimber, respectively. Variability in stumpage values results not only because of changes in annual market conditions due to demand factors such as housing starts, the economy, etc., but also annual differences across the State in the particular characteristics of what stand types are actually harvested (e.g., thinnings versus final harvests) and localized markets since these prices are statewide averages. Thus, one final set of revenues for each product class was examined. These were weighted average prices by harvest amount from each year using reported revenues from years 2011 to 2014 (inflation was ignored). Weighted revenues were \$16.90, \$25.14, and \$35.25 per cord for pulpwood, bolt and pulpwood, and sawtimber, respectively.

When using 2011 prices, there are differences in the optimum regimes between 2012 prices (fig. 6). For the low quality site, both the 150_90 and 120_90 regimes produce similar economic return (\$-256.72 at age 60) because the regimes didn't differ in terms of operations until age 60 (fig. 2). For the high quality site, the optimum treatment became 150_120 at age 50 (\$-174.84). However, for both sites, treatments are similar, with the only real exception being the 120_60 treatment. Interestingly, 2011 prices show a decrease in revenues; this is likely due to much lower sawtimber revenues (\$40.46 in 2012 versus \$32.12 in 2011).

The weighted revenues also showed the greatest economic return on the low quality site occurs with the 150_90 or 120_90 treatments (-\$224.45), with the optimal rotation age being 60 (fig. 7). On the high quality site, the optimal scenario is 150_120 treatment (-\$128.79). This treatment may be difficult to implement operationally, and hence not thinning at all or the 150_90 treatment may be the best alternative operationally. Of the three sets of revenues, the weighted has the highest revenues for both the bolt and pulp class and the pulpwood only class.

The DNR management guide recommends not to thin on sites with SI values less than 60 feet, with

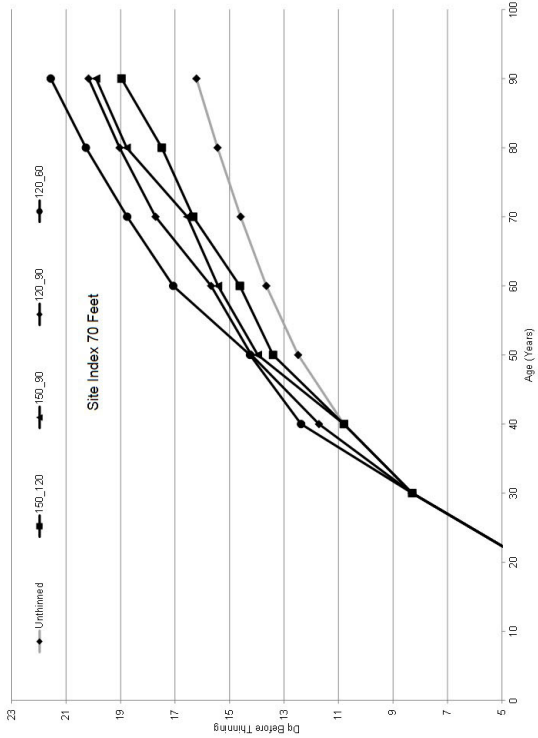
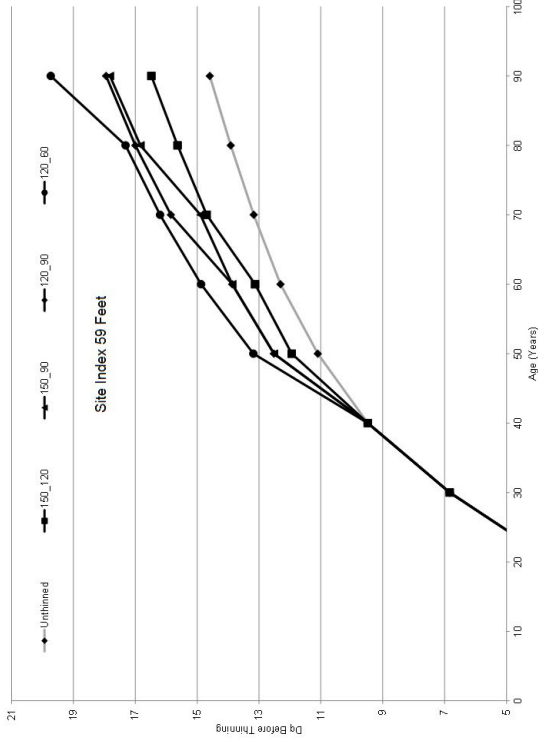


Figure 4—Quadratic mean diameter (Dq—inches) by treatment prior to thinnings. Thinning treatments are defined in table 1.

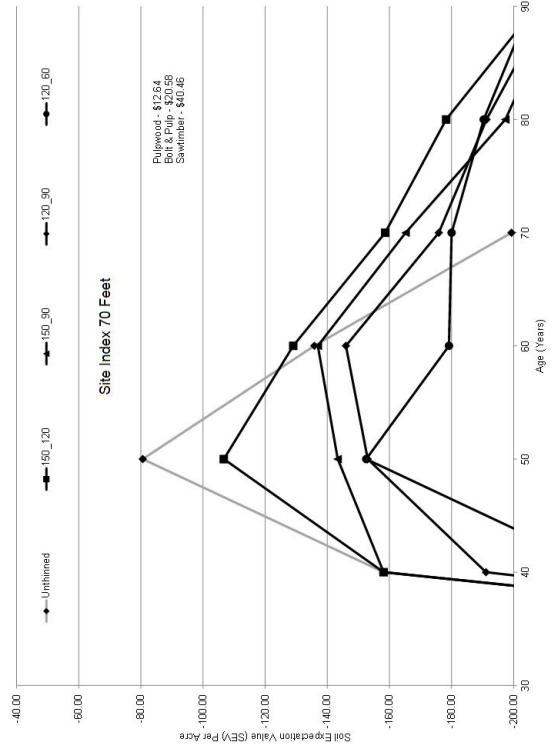
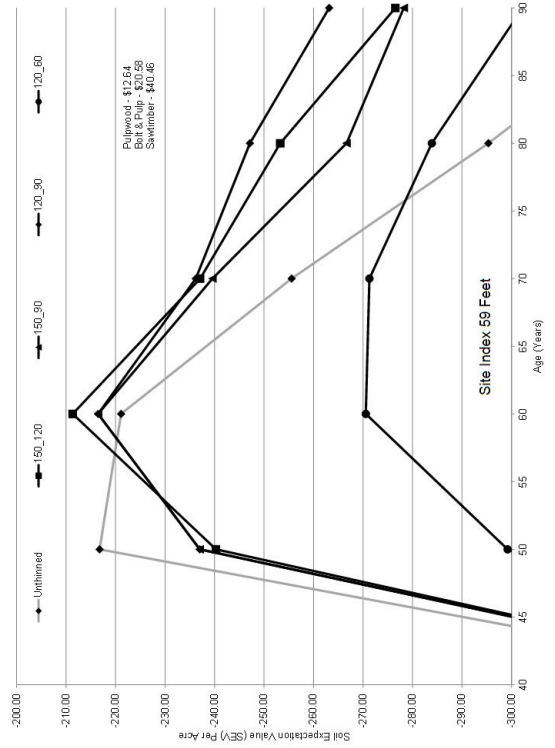


Figure 5—Soil Expectation Value (SEV) by treatment. Thinning treatments are defined in table 1.

thinning on higher quality sites considered optional. In terms of economics, LS results are consistent with these guidelines. Thinning did not appear to substantially increase economic returns and could actually decrease returns (figs. 5 to 7). According to LS, though, thinning increases cumulative merchantable volumes (fig. 3). However, these results assume pure white spruce plantations, when in fact most, if not all, DNR stands have a mix of species because they are currently being managed more for multiple objectives.

From a purely economic perspective and given current markets, perhaps it is best not to thin high quality sites. On low quality sites, it appears that the 150_120 treatment maximizes returns. However, there are minor differences between the unthinned, 150_90, and 150_120 scenarios for both high and low quality sites that are probably academic given “real world” markets, “local” markets (e.g., a plantation located near Grand Rapids and hence UPM Blandin Paper Company may have higher pulpwood stumpage prices thus justifying planting), harvesting costs, actual plantation planting schemes, “real world” vegetation control, etc. These findings are somewhat consistent with the DNR management guide, which states for SIs less than 60 feet, thinning should not be conducted (pulpwood rotation) and on sites of 60 feet and greater thinning can or cannot be conducted. Generally though, on higher quality sites, thinning would likely be beneficial, especially when considering factors such as earlier returns to help reduce financial risk and when considering forest health.

Number and Predicted Yields of Thinnings

Thinnings first occurred at age 40 in all regimes for the lower quality site, and for the higher quality site at age 30 for the 120 target basal area and at age 40 for the 150 target basal area (fig. 2). Projections were on a 10-year interval; first thinnings may have occurred at ages of 25 or 35 if run on a 5-year interval.

As expected, the number of thinnings differed by treatment. On the low quality site, two thinnings occurred for the 120_60 (ages 40 and 80) and 150_90 (ages 40 and 70) regimes, while three thinnings occurred for the 120_90 and 150_120 (ages 40, 60, and 90) regimes. On the high quality

site, two thinnings occurred for the 120_60 (ages 30 and 50) and 150_90 (ages 40 and 70) regimes, three thinnings occurred for the 150_120 (ages 40, 60, and 80) regime, and four thinnings occurred for the 120_90 (ages 30, 40, 60, and 90) regime. Within many recent DNR landscape modeling efforts, for plantations of any site quality, up to two thinnings can occur (after age 25 with a minimum 15-year interval between thinnings) where each generates 10 cords per acre. Currently, Minnesota Forest Industries (MFI) assumes two thinnings as well will occur where the first generates 10 cords per acre (must occur between ages 30 and 35), and the second 12 cords (must occur between ages 40 and 50).

On the low quality site, for the 120_60 regime, thinning cords were both 24 per acre; for the 120_90 regime, thinning cords ranged from 12 to 17 averaging 15 cords per acre; for the 150_90 regime, thinning cords were 17 and 21 averaging 19 cords per acre; while for the 150_120 regime, thinning cords ranged from 10 to 13 averaging 12 cords per acre. On the high site, for the 120_60 regime, thinning cords were 16 and 23 averaging 19 cords per acre; for the 120_90 regime, thinning cords ranged from 10 to 17 averaging 13 cords per acre; for the 150_90 regime, thinning cords were 26 and 24 averaging 25 cords per acre; while for the 150_120 regime, thinning cords ranged from 14 to 18 averaging 15 cords per acre.

In southeastern Ontario, Stiehl (1970) reported thinned cords obtained through thinning from below treatments ranged from 3 to 12.5 cords per acre during first and second thinnings for ages ranging from 32 to 44 and SI values near 65 feet (base age 50). Stiehl (1980) reported first and second thinned merchantable cords ranging from 7 to 11 cords per acre (although one observation had a value of only 2 cords—most likely an outlier) at ages 33 and 43. Sites had site indexes near 60 (near the low end of DNR “operational” thinned stands), and since the second thinning occurred 10 years after the first as opposed to 15 years after the first, a simple assumption of 10 cords per acre within DNR landscape modeling assessments seems reasonable.

Wilde and others (1965) report a standing basal area per acre of 136 square feet and a standing volume of 36 cords per acre in a 33-year-old plantation

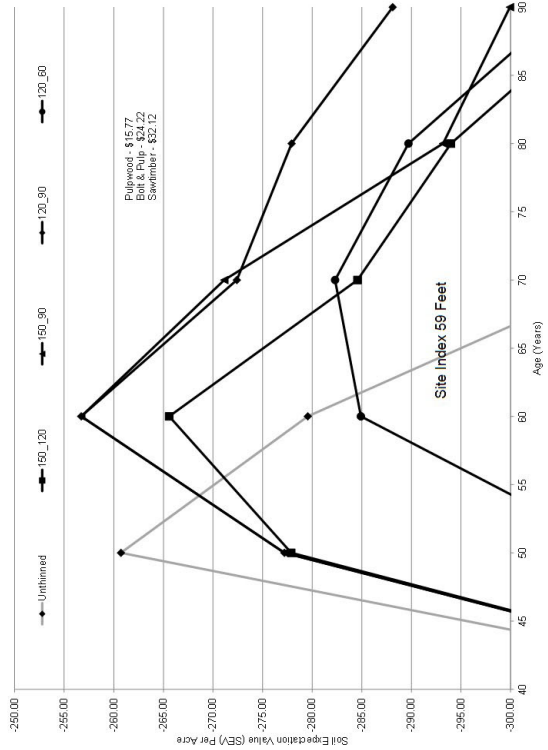
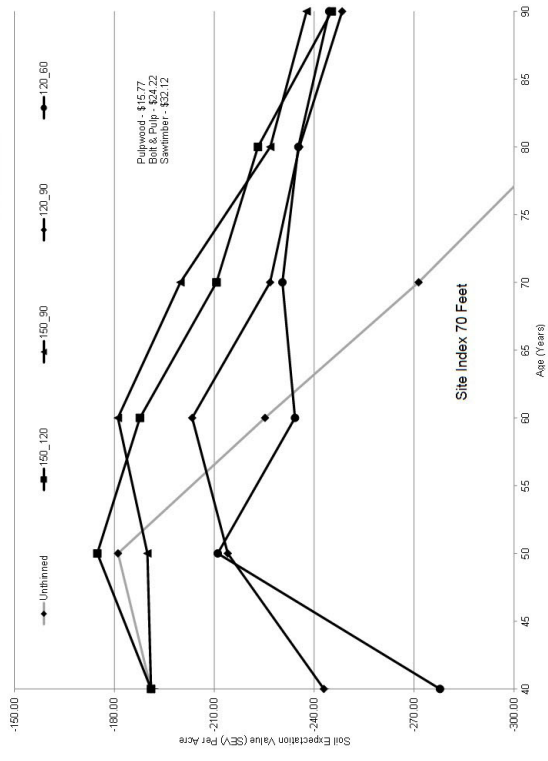


Figure 6—Soil Expectation Value (SEV) by treatment. Thinning treatments are defined in table 1.

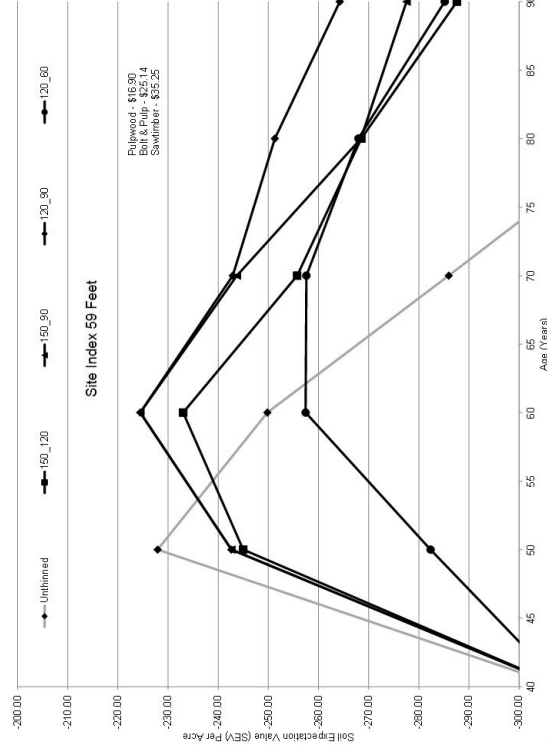
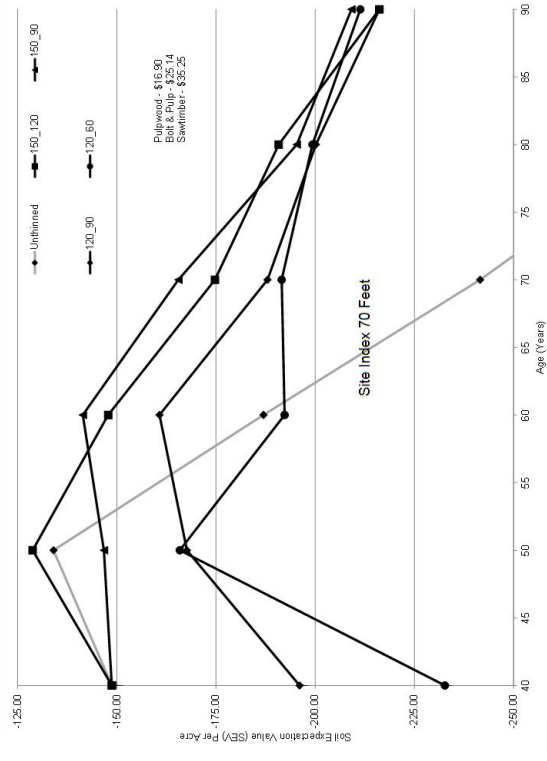


Figure 7—Soil Expectation Value (SEV) by treatment. Thinning treatments are defined in table 1.

in Wisconsin, an every third-row thinning would essentially reduce basal area to around 90 square feet and remove around 12 cords per acre. D'Amato and others (2011) reported pre-thinning basal area per acre observations from northern Minnesota plantations ranging in age from 25 to 46 years old, site indexes (base age 50) from 38 to 77 feet, and planting densities ranging from 600 to 1,300 seedlings per acre. Based on a simple assumption of 0.7080 cords per square meter of basal area (based on Stiell 1980), and assuming a third-row thinning, around 4 to 13 cords per acre would be generated, with an average of 9.42 cords per acre.

Optimum Residual Stand Density to Maximize Cumulative Volumes

Several assessments of optimum plantation thinning regimes have been conducted. Fifteen years after thinning a 23-year-old plantation in Wisconsin (planted at a density of 5,400 seedlings per acre and thinned from below), Wambach and Cooley (1969) found that lower residual densities increased average tree diameter. They also found that 15-year volume increment was greatest at a residual basal area ranging from around 100 to 120 square feet per acre. Stiell (1980) conducted a thinning experiment in an old-field with a site index of 59 feet (base age 50), planted to a spacing of 1,582 seedlings per acre, and first thinned at age 33. Three thinnings were conducted, at ages 33, 43, and 53. At age 53 (prior to the 3rd thinning), he found that a residual thinning density of 140 square feet per acre produced more cumulative (standing plus cut) total and merchantable volume than residual densities of 80 and 110 square feet and an unthinned treatment in a southeastern Ontario plantation. A residual density of 80 square feet produced the lowest cumulative volumes.

From a cumulative volume production perspective at commonly used rotation ages (e.g., 50 to 90 years), this study using LS basically recommends thinning at a target basal area of 150 square feet. Thinning to a residual basal area of 90 or 120 square feet did not produce substantial differences in cumulative volume. Most likely, thinning to a residual basal area of 90 square feet per acre will be best operationally and is consistent with the Wisconsin DNR management recommendations; thin at 160 square feet back to a residual basal area of 90 square feet per acre at the first thinning, with

possible future thinnings to a residual basal area of 90 (generally in pole-sized material) or 120 square feet per acre (generally in sawlog-sized stands).

CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

On higher quality sites (i.e., SI greater than 60 feet), the greatest economic return appears to be when light but frequent thinnings occur (150_120) with the optimal rotation age being around 50 years (figs. 5 to 7). Not thinning at all would be a financial alternative. The 150_90 scenario is relatively competitive with an optimal rotation age of around 60. Operationally, due to the time and costs associated with thinnings, the 150_90 scenario may be more practical to implement. On lower quality sites, the 150_90 scenario appears to be best financially with the optimal rotation age of 60. Not thinning is also a viable alternative with a rotation age of around age 50.

From a cumulative volume production perspective at commonly used rotation ages (e.g., 50 to 90 years), this study using LS basically recommends thinning at a target basal area of 150 square feet. Thinning back to 90 square feet or 120 square feet didn't seem to produce much difference in cumulative volume. Most likely thinning back to 90 square feet will be best operationally.

Consistent with current practice, thinnings should begin around ages 30 to 40, and likely at age 25 on better sites. When using economic rotation ages of either 50 or 60 years on the lower quality site, likely only one thinning can be conducted around age 35 or 40. On higher quality sites, if thinnings are implemented, one or two thinnings could be conducted depending on the target basal area of either 150 or 120 square feet, respectively. The first thinning should be conducted around age 25 to 35 and the second around age 40 or 45. Not thinning appears to be a financially viable alternative, but concerns about disease may necessitate thinning. This analysis suggests that from an economic perspective and given current markets and costs and the growth rates associated with LS, justification for white spruce plantations across the State as a whole would need to include some consideration of amenity values (e.g., landscape cover type diversity, wildlife habitat, employment considerations, etc.).

ACKNOWLEDGMENTS

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Comparing Unthinned Slash Pine Plantation Yield Predictions From Time-of-Planting

Curtis L. VanderSchaaf, Gordon Holley, and Joshua Adams¹

Abstract—Slash pine (*Pinus elliottii* Engelm.) has been planted in the Western Gulf region, but it has been studied less extensively than other southern yellow pines. Several yield prediction systems, including the Forest Vegetation Simulator (FVS), have been used to examine how different management options likely impact financial returns of these plantations. The objectives of this study were to compare projections from FVS to two other publicly available growth and yield systems in the Western Gulf region named SLAeatx and COMPUTE P-SLASH (CSLASH).

Predictions from the time-of-planting were obtained for densities of 300, 500, and 700 stems per acre for site indexes of 50, 70, and 90 feet (base age 25). CSLASH was developed based on observations of plantations in Louisiana, Mississippi, and Texas, while FVS projections were modified to best represent conditions in central Louisiana. The SLAeatx program is based exclusively on observations obtained from East Texas plantations.

From ages of 10 to 20, fairly consistent projections of basal area and merchantable volume for all nine combinations of planting densities and site qualities were observed. Within the range of common rotation ages (e.g., 15 to 30 years), projections of both basal area and merchantable volume in some cases differed substantially. SLAeatx consistently underpredicted volume within this range of ages. At older ages, FVS greatly exceeded SLAeatx, and also CSLASH for lower site qualities.

INTRODUCTION

Slash pine (*Pinus elliottii* Engelm.) is native to the Southeastern United States and has been planted in the Western Gulf region. Economic rotation ages range from 20 to 40 years depending on site quality, management, and markets. According to a recent Forest Inventory and Analysis (FIA) estimate (Miles 2016), there are 446,605, 189,159, and 73,552 acres of plantations in Louisiana, Mississippi, and Texas, respectively (fig. 1). Tools are needed to examine how different management options likely impact financial returns. Several yield prediction systems have been developed to estimate future yields of these plantations. These systems allow users to examine how different planting densities, site qualities, and financial assumptions impact unthinned stand development, rotation age, and financial returns.

The objectives of this study were to compare projections of unthinned stands from the Forest Vegetation Simulator (FVS) to two other publicly

available growth and yield systems in the Western Gulf region—SLAeatx and CSLASH.

METHODS

Forest Vegetation Simulator (FVS)

The Southern variant (SN, version 1860) of FVS covers forest areas in the Southern United States including Louisiana, East Texas, and Mississippi (Dixon 2002, Keyser 2008). SN model relationships were fit in the early 2000s using FIA periodic inventory data from all Southern States. For slash pine, enough data existed to modify growth of plantations. This is a distance-independent individual tree model.

Within SN, the “Bareground” option was used to generate plantations of 300, 500, and 700 seedlings per acre. Survival at age 1 was assumed to be 100 percent, and the “Sprouting” option was turned off to eliminate natural regeneration. The “Managed” keyword was used to reflect that in general

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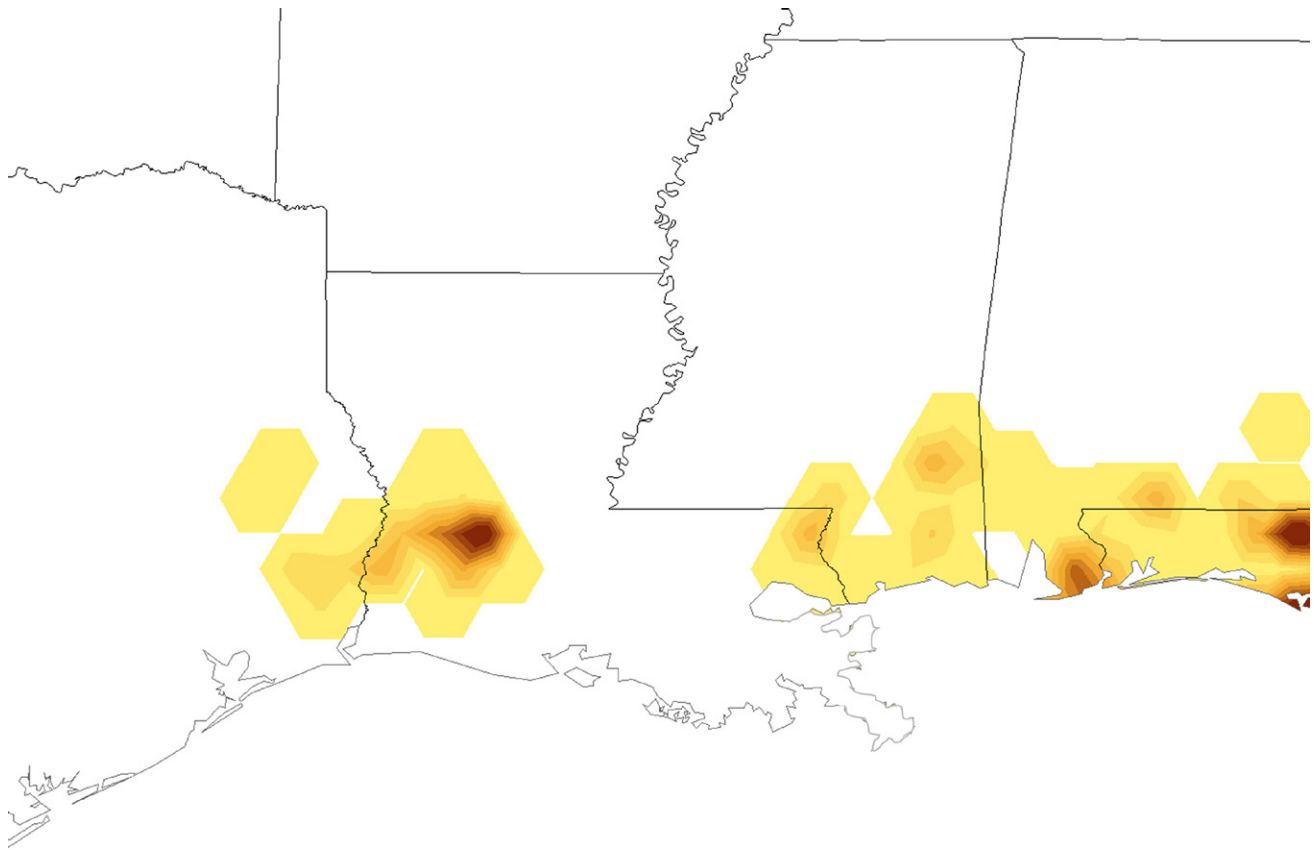


Figure 1—Spatial distribution of slash pine plantations based on interpolating among USDA Forest Service Forest Inventory and Analysis (FIA) plots when using standing volume (VOLCFNET in FIA). Spline with tension using a maximum of 16 nearest neighbors. A darker shade represents a greater relative concentration of slash pine volume.

plantations have greater diameter growth rates relative to natural or “unmanaged” stands.

Minimum merchantability limits were consistent with standard FVS SN protocol, and stump height was set to 1 foot. Minimum merchantable pulpwood diameter at breast height (4.5 feet about ground level, DBH) was 4.0 inches, and upper stem diameter inside-bark (DIB) was 4.0 inches. The default FVS max Stand Density Index (SDI) of 435 was used.

Site index equations within FVS use a base age of 50. However, projections were desired based on site indices using a more common base age of 25. Hence, site indices (base age 50 year) of 70 feet and 99 feet were specified within FVS to produce site indices of 50 feet and 70 feet (base age 25 year), respectively. A site index of 90 feet at base age 25 could not be produced since the maximum site

index within FVS is 105 feet (base age 50), which equates to a site index of 74 feet (base age 25). FVS keyword “StdInfo” was used to specify the stand being located in the Kisatchie National Forest (Kisatchie).

SLAEATX

Data used in developing equations (Coble and Lee 2008, Coble 2009) were obtained from long-term measurements of operationally established unthinned plantations across the growing conditions of East Texas as part of the East Texas Pine Plantation Research Project—ETPPRP (<http://www.faculty.sfasu.edu/cobledean/ETPPRP.html>). A total of 84 plots were established ranging in planting density from 453 to 1,361 seedlings per acre. Plantations represented by these plots ranged in total age from 2 to 33 years, 60 to 1000 trees per acre, and 35 to 100 feet site index (base age

25). Site preparation was minimal with the most intensive consisting of raking, piling, and burning. The plantations were established using bareroot seedlings and woods-run genetic stock, and they were established prior to 1980. As a result, the regeneration practices and seedling quality (through improved nursery practices) are not necessarily indicative of more recent regeneration practices (e.g., intensive management, and Elite or Mass Control Pollinated seedlings [MCP]). SLAeatx is a diameter-distribution model.

Merchantability specifications were the same as those used by FVS with the exception that an upper stem 4.0-inch diameter outside-bark (DOB) was used. Merchantable green tons per acre (all trees were assumed to be in the pulpwood class) were converted to merchantable cubic feet assuming 69 pounds per cubic foot of wood and bark.

Cutover Slash Growth and Yield Model— CSLASH

Data used in developing equations were obtained from 507 unthinned-stand yield observations and 543 thinned-stand growth-period observations in operational plantations established on problem-free cutover forest land in Louisiana, Mississippi, and Texas (Zarnoch and others 1991). The plots were not located in areas where survival was poor or where heavy insect, disease, or other damage was present. Ages of plantations ranged from 8 to 47 years old, planting density ranged from 251 to 1750 seedlings per acre, and site index ranged from 28 to 87 feet (base age 25).

Within CSLASH, a 1-foot stump and a log/bolt length of 4 feet were specified. CSLASH is a diameter-distribution model.

To be as consistent with FVS as possible when comparing projected volumes, a minimum 4.0-inch DBH and a 4.0-inch DOB were used in both SLAeatx and CSLASH.

Projections

Projections were obtained for each system if applicable for planting densities of 300, 500, and 700 stems per acre for site indexes of 50, 70, and 90 feet (base age 25). Hence, there was the potential for nine model runs per system. However, as

mentioned before, for FVS a site index of 90 feet at base age 25 could not be produced, and thus for that system only six model runs were conducted.

A site index of 90 feet is certainly at the extreme. For the data used in developing CSLASH (only unthinned), only 3 percent (14 of 507) of the plantations had site indexes (base age 25) of roughly 80 feet and greater. However, for the ETPPRP dataset, the maximum site index reported was 100, and the average was 73.7 feet with a standard deviation of 12.2 (Coble and Lee 2008). Hence, there appears to be a fair amount of stands with site indexes of 80 feet and greater. Nonetheless, caution should be used when applying any of these projection systems to similar plantations.

RESULTS AND DISCUSSION

There were some meaningful differences between the three model systems (figs. 2 and 3). From ages of 10 to 20, fairly consistent projections of basal area and volume for all nine combinations of planting densities and site qualities were observed. However, at the extremes, volume projections were vastly different. Within the range of common rotation ages (e.g., 15 to 30 years) projections in some cases differed substantially. SLAeatx consistently underpredicted volume within this range of ages. At older ages, FVS greatly exceeded SLAeatx, and also CSLASH for lower site qualities. CSLASH had much higher volume predictions relative to SLAeatx on the site index 90 site. Total merchantable volumes of 8,000 to 12,000 cubic feet seem excessively high and greatly exceed published amounts (e.g., Bennett 1963, Pienaar and others 1996, Dickens and Will 2004, Scott and Tiarks 2006).

For a site index of 50, SLAeatx reached basal area asymptotes sooner than FVS and generally at lower levels for a particular planting density and site quality combination, while for a site index of 70 the two systems produced asymptotes around the same age. For basal area (e.g., at DBH), FVS predicted basal area begins at an earlier age relative to SLAeatx. SLAeatx generally predicted the first occurrence of merchantable volumes at younger ages. However, the earliest possible projection age in CSLASH is 12 years old.

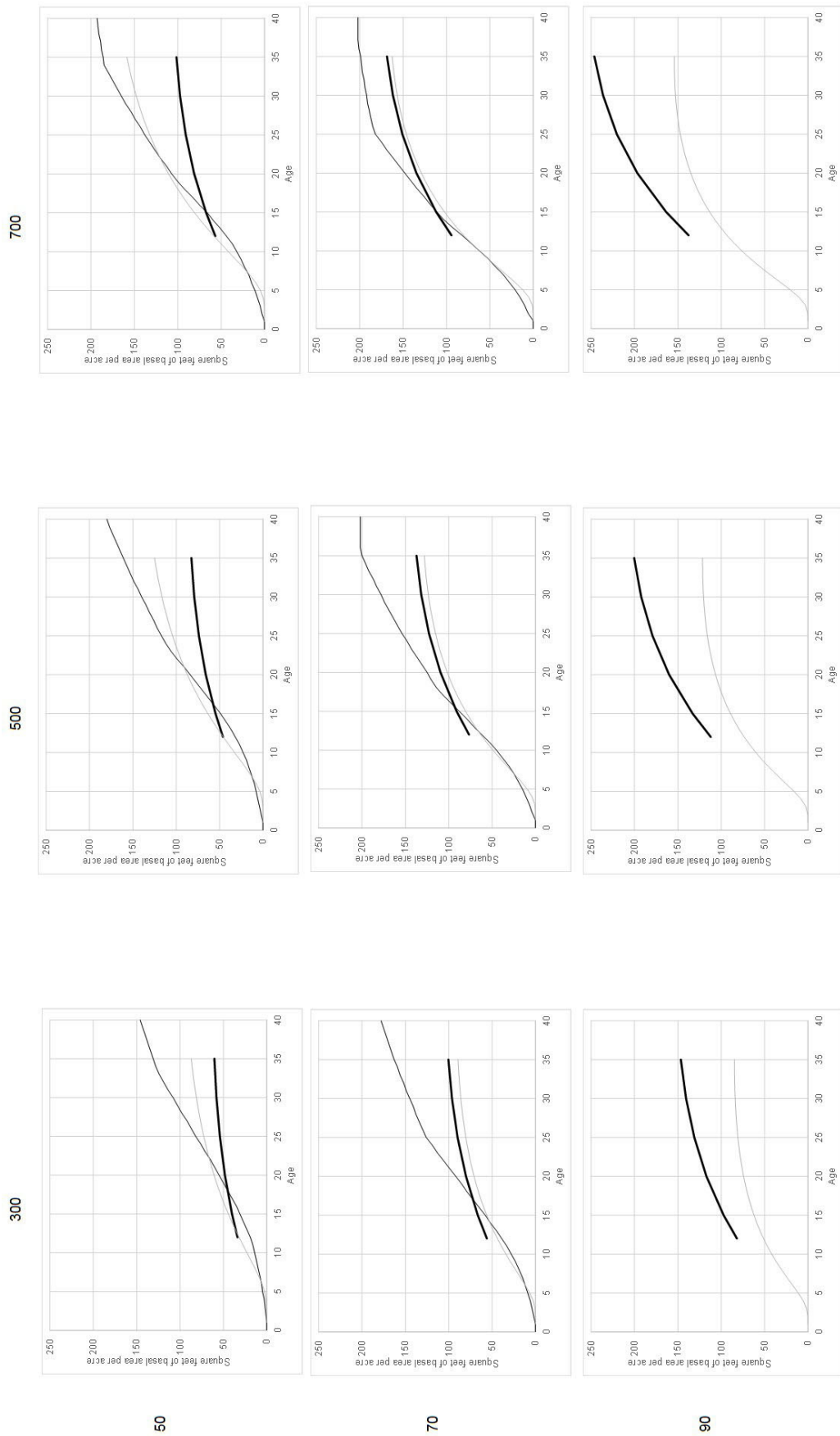


Figure 2—Basal area projections for three model systems by planting density (300, 500, and 700 seedlings per acre) and site index (50, 70, and 90 feet—base age 25). The black bold lines are projections from CSLASH, gray lines are from SLAeatx, and black lines are from FVS. Projections for a site index of 90 feet (base age 25) could not be produced within FVS.

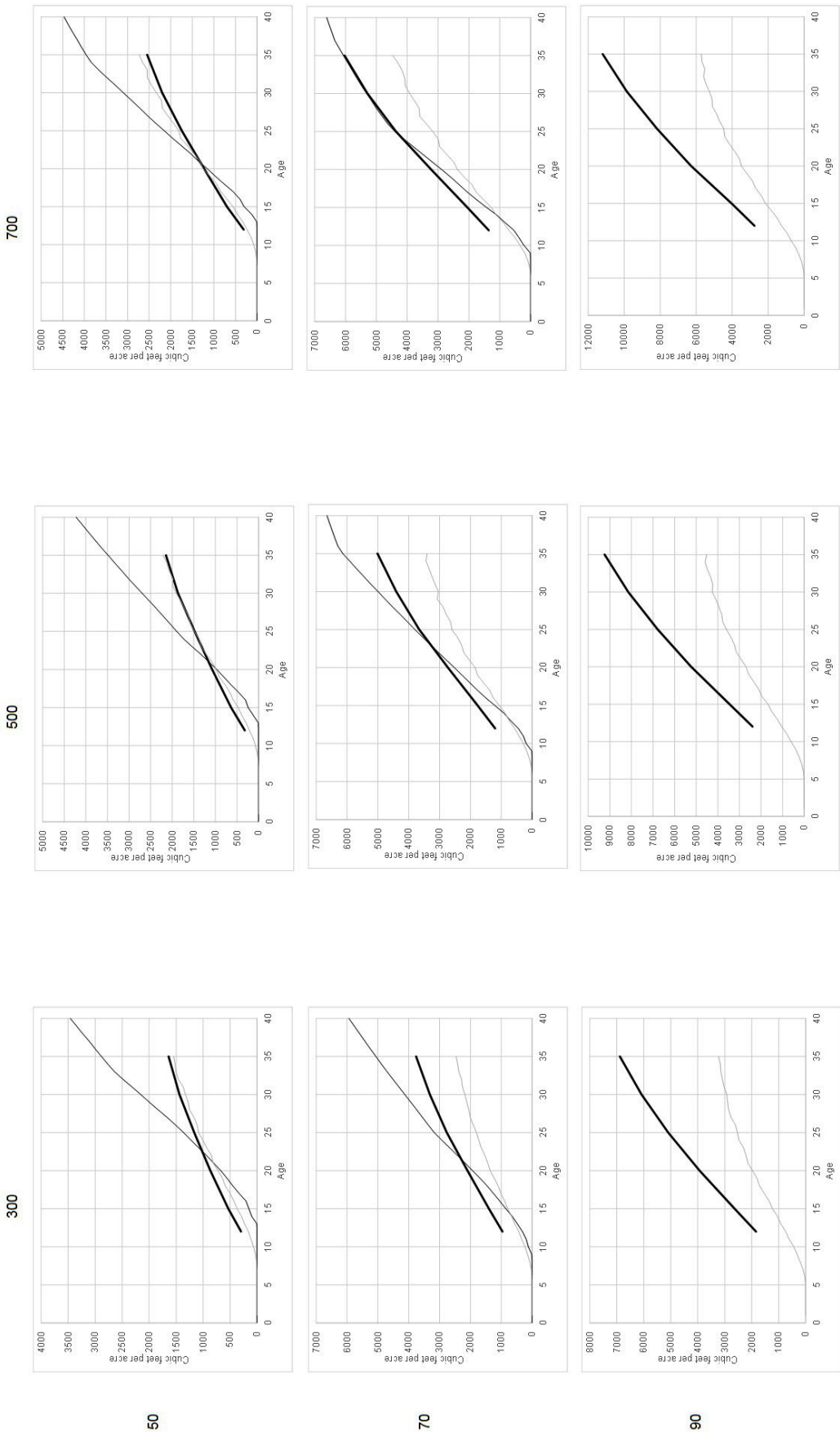


Figure 3—Cubic foot volume projections for three model systems by planting density (300, 500, and 700 seedlings per acre) and site index (50, 70, and 90 feet—base age 25). The black bold lines are projections from CSLASH, gray lines are from SLAeatx, and black lines are from FVS. Projections for a site index of 90 feet (base age 25) could not be produced within FVS. Cubic foot volume is total merchantable volume.

A simple validation analysis was conducted by comparing projections from the three model systems with observed results found in Ferguson and Baldwin (1995) for a study established near Woodworth, LA. Based on dominant height data (10-T in their table 1 across all planting densities and for ages 6, 10, and 15) presented in the paper, a simple anamorphic site index curve was developed. Dominant height data at age 15 years along with the site index equation was used to project forward to a base age of 25 years, producing a site index estimate of 66 feet. This site index was used in all three projection systems. To obtain a site index of 66 feet at base age 25, within FVS a value of 95 (base age 50) was used.

SLAeatx only presents estimates for planting densities ranging from 300 to 1,400 seedlings per acre. Hence only planting densities of 1,210, 908, 681, and 436 were compared. It is not known for sure whether data from the plantations summarized in Ferguson and Baldwin (1995) were used in developing CSLASH. Reported values in table 1 of Ferguson and Baldwin (1995) are based on the average of five replications. Hence, results are indicative of the likely ability of the three projection systems to predict future slash pine plantation development. Only basal area is compared since Ferguson and Baldwin (1995) reported total (not merchantable) cubic foot volume.

All three models produced reasonable estimates of basal area for all planting densities within the range of observed stand ages (fig. 4). CSLASH slightly overpredicted for the higher planting densities. However, all three slightly underpredicted basal area for the two lower planting densities.

CONCLUSIONS

In some cases, basal area and volume within the range of common rotation ages (e.g., 15 to 30 years) were substantially different among the three systems. CSLASH had much higher volume predictions for greater site qualities. At older ages, predicted volumes from FVS greatly exceeded SLAeatx, and also CSLASH on lower site qualities.

Although not presented in this paper, economic projections for extreme combinations of planting density and site qualities using CSLASH should

be viewed with caution. Merchantable volumes appear to be excessively high in some cases. However, for ages between 15 and 30 years and moderate combinations of planting density and site qualities, CSLASH appeared to provide reasonable projections. SLAeatx underpredicted relative to the two other systems and thus financial assessments may be conservative.

SLAeatx and CSLASH are both diameter-distribution models, while FVS is a distance-independent individual tree model. However, the model construction type doesn't seem to produce a consistent impact on the projections. Most likely differences in the data used to develop the systems produced variability among predictions more so than the model type. CSLASH was developed using data obtained from "problem-free" sites. The ETPPRP plantations (SLAeatx) did have a fair amount of fusiform rust (*Cronartium quercuum* f.sp. *fusiforme*) infection that likely impacted yields. These infections may have resulted in lower predicted yields relative to CSLASH and FVS.

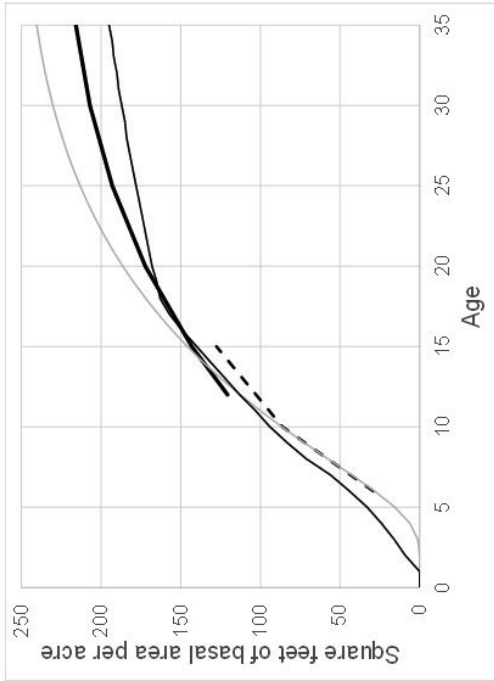
ACKNOWLEDGMENTS

We would like to thank Michael Blazier and David South for providing useful comments.

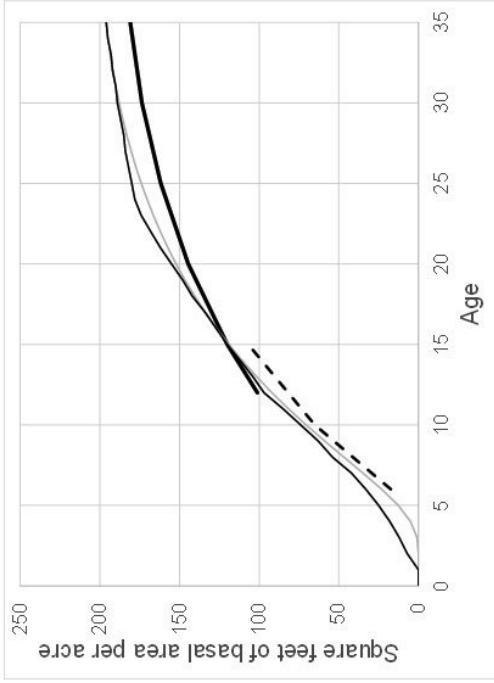
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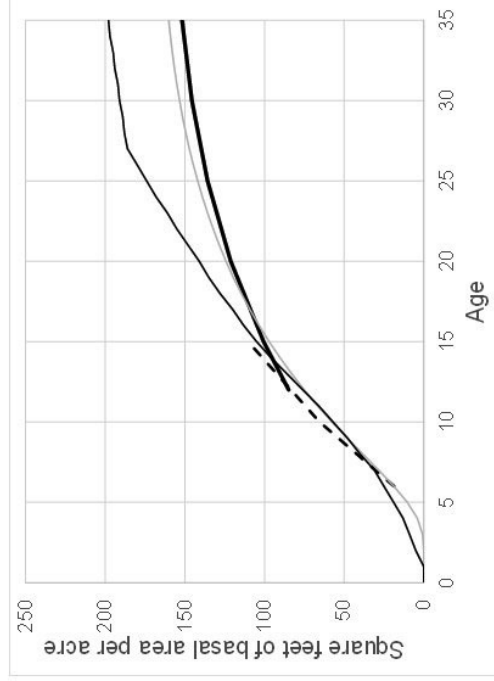
1,210



908



681



436

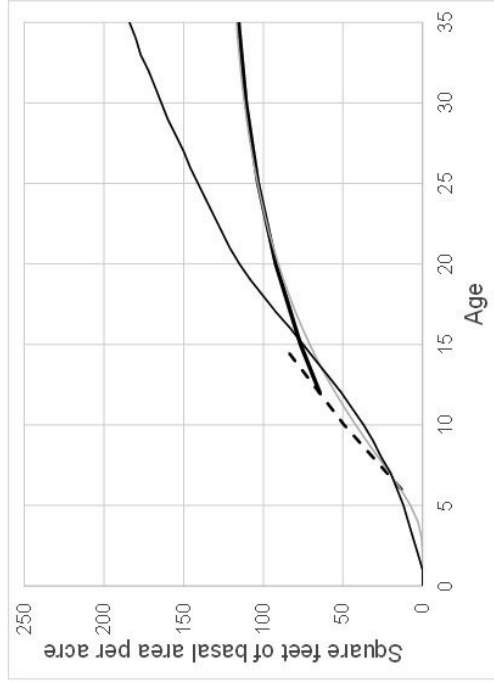


Figure 4—Basal area projections for observed data from Ferguson and Baldwin (1995) and three model systems by planting density (436, 681, 908, and 1,210 seedlings per acre) with a site index of 66 feet (base age 25). The dark dashed line are observations from Ferguson and Baldwin (1995), black bold lines are from CSLASH, gray lines are from SLAeatx, and black lines are from FVS.

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Even- and Uneven-Aged Management Scenarios for Maximizing Economic Return in the Sweetgum-Nuttall Oak-Willow Oak Bottomland Hardwood Forest Types in the Lower Mississippi Alluvial Valley

Sunil Nepal, Brent R. Frey, and James E. Henderson¹

Abstract—A fully-stocked, naturally regenerated sweetgum-Nuttall oak-willow oak stand was modeled under an even-aged and 27 uneven-aged management scenarios to maximize Net Present Value (NPV). The uneven-aged scenarios were implemented using single-tree selection governed by the BDq approach, with uneven-aged scenarios representing a range of target stand conditions (i.e., different maximum diameters, q-factors, and regeneration levels). The Forest Vegetation Simulator (FVS) was used to model growth and yield under the different scenarios. Average 10-year price data was used for valuation of yield data under the different scenarios, and a 3-percent discount rate was applied. Even-aged management produced higher overall NPVs in all but two scenarios. Generally, as the q-factor increased, the tradeoff between even- and uneven-aged management decreased. Likewise, as the maximum diameter target was reduced, the tradeoff between even- and uneven-aged management decreased, except for the highest q-factor. Among the uneven-aged management scenarios, medium regeneration scenarios produced higher NPV than the lower and higher regeneration scenarios. Results of this study will provide a useful management tool to compare the tradeoff of even-aged management versus uneven-aged management in the sweetgum-Nuttall oak-willow oak forest type for a range of management scenarios.

INTRODUCTION

The sweetgum-Nuttall oak-willow oak forest type is one of the most ecologically and commercially important forest types in the Lower Mississippi Alluvial Valley (LMAV), representing approximately 17 percent of the total forested area (Oswalt 2013). This forest type is mainly composed of sweetgum (*Liquidambar styraciflua* L.), Nuttall oak (*Quercus nuttallii* Palmer), willow oak (*Quercus phellos* L.), cherrybark oak (*Quercus pagoda* Rafinesque), water oak (*Quercus nigra* L.), and green ash (*Fraxinus pennsylvanica* Marsh.) (Meadows and Stanturf 1997). It is one of the most highly valued bottomland forest types because it produces high quality hardwood timber (Meadows and Hodges 1997), provides ideal habitat for several priority wildlife species (Twedt and others 2012) and myriad other ecosystem services. Forest management practices can differ depending upon landowner objectives for timber, wildlife habitat, aesthetics, or other. For example, timber-centric management favors optimization of timber

revenue, which is considered to be more efficiently produced through even-aged management due to the shade intolerance of most desirable southern hardwood timber species. In contrast, wildlife-focused management prioritizes forest conditions with greater structural diversity, which are often thought to be best achieved through uneven-aged management. Tradeoffs between even- and uneven-aged management can be examined by economic valuation of timber harvests for each management scenario. We expect that for some stand conditions the economic tradeoff may be negligible, while for other stands it may be substantial. In either case, this approach can help guide landowners and managers in understanding the timber-based valuation tradeoff that can result from favoring even- or uneven-aged management.

While economic valuation is needed to help decisionmakers make informed choices regarding alternative management approaches, little economic analysis has been done on bottomland hardwood

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forest management. Nepal and others (2016) examined economic returns of even- and uneven-aged management in the sweetgum-Nuttall oak-willow oak forest type and found that an even-aged management scenario outperformed an uneven-aged management scenario. They simulated 34 different sweetgum-Nuttall oak-willow oak stands; both even- and uneven-aged scenarios were managed to optimize NPV of timber harvests. Although, they simulated a wide range of initial stand conditions (i.e., 34 different stands), only one stand structural target and regeneration condition was evaluated for uneven-aged management (i.e., 1.3 q-factor, 38-inch maximum residual DBH, and 104 seedlings/ acre/cutting cycle). We hypothesized that different stand structural target conditions and regeneration parameters could potentially generate different returns. To examine the tradeoff between optimal even-aged management and uneven-aged management, we compared multiple uneven-aged management scenarios to an optimal even-aged management scenario and estimated the economic tradeoff between even- and uneven-aged management. We chose a fully-stocked sweetgum-Nuttall oak-willow oak inventory plot from the U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis (FIA) database. Twenty-seven hypothetical scenarios were created in uneven-aged management, and an optimal condition was identified for each scenario. Growth and yield produced by each scenario in combination with timber price data were used to estimate economic returns from each scenario and evaluate tradeoffs between even- and uneven-aged management.

METHODS

Stand Simulation

A fully stocked plot from the FIA database was selected. This stand was chosen because it was an average stand in terms of species composition, maturity, stocking, and site quality. The stand was estimated to be 27 years old, with standing basal area of 111 square feet per acre, 781 trees per acre, standing volume of 2,382 cubic feet per acre, and stocking (based on Goelz 1995) at 74.5 percent. The Southern Variant (SN, version 1943) of the Forest Vegetation Simulator (FVS) was used to model the simulations. SN includes a partial establishment model which does not automatically predict natural regeneration other than sprouting from

stumps and root suckers following cutting/fire stem mortality (Dixon 2002, Keyser 2008). Thus, natural regeneration was estimated based on the average regeneration density of the sweetgum-Nuttall oak-willow oak forest type according to FIA plot data available for the same ecological range in the LMAV area (see Nepal and others 2016). Average regeneration included 18 different species, with hackberry-sugarberry (*Celtis* spp.), cherrybark oak-Nuttall oak, and green ash seedlings predominating (table 1). Site index of the selected stand was 97 feet for sweetgum (base age 50).

Management Scenarios

Given the existing stand conditions, the management scenarios required different modeling approaches to achieve continuous sustained yields, which was a requirement of the economic comparison. In the even-aged scenario, the existing stand was managed based on the decisionmaking criteria for managing bottomland hardwood stands described by Goelz and Meadows (1997) to maximize NPV. After harvesting the existing stand, a second rotation was assumed to be repeated in perpetuity to maximize Land Expectation Value (LEV). Thinning treatments were applied based on the bottomland hardwood stocking guide (Goelz 1995). When stand stocking reached the 100-percent stocking (A-line) level on the stocking guide, the stand was thinned to the B-line level of stocking. Species preference was given to the oak component, so that in each thinning inferior quality and less valuable hardwood species were removed favoring the retention of oak species to the end of the rotation.

The uneven-aged management scenario followed the BDq approach using single tree selection (O'Hara and Gersonde 2004). Twenty-seven hypothetical scenarios were developed based on a range of q-factors, maximum residual DBHs, and regeneration densities. This included three q-factors (1.3, 1.4, and 1.5), three maximum residual DBH (24, 30, and 38 inches), and three regeneration densities of low, medium, and high (10, 20, and 100 percent of the average regeneration calculated in table 1). We set diameter class width as two during the simulation process. Based on Putnam and others (1961), a desirable post-harvest target residual basal area of 68 square feet per acre was applied in uneven-aged scenarios. Several cutting cycles were

Table 1—Calculated average regeneration density per acre in the sweetgum-Nuttall oak-willow oak forest type in the Lower Mississippi Alluvial Valley based on the FIA data

Species	Scientific name	Alpha code ^a	FIA code ^a	Regeneration density/acre
Redcedar species	<i>Juniperus</i> spp.	JU	57	6
Boxelder	<i>Acer negundo</i>	BE	313	6
Red maple	<i>Acer rubrum</i>	RM	316	38
American hornbeam	<i>Carpinus caroliniana</i>	AH	391	8
Hickory species	<i>Carya</i> spp.	HI	400	36
Hackberry-sugarberry	<i>Celtis</i> spp.	HB	460	226
Common persimmon	<i>Diospyros virginiana</i>	PS	521	23
Green ash	<i>Fraxinus pennsylvanica</i>	GA	544	156
Sweetgum	<i>Liquidambar styraciflua</i>	SU	611	50
Cherrybark oak-Nuttall oak	<i>Quercus pagoda</i> - <i>Q. nuttalli</i>	CB	813	184
Overcup oak	<i>Quercus lyrata</i>	OV	822	26
Water oak-willow oak	<i>Quercus nigra</i> - <i>Q. phellos</i>	WK	827	89
Post oak	<i>Quercus stellata</i>	PO	835	6
Willow species	<i>Salix</i> sp.	WI	920	18
Winged elm	<i>Ulmus alata</i>	WE	971	75
American elm	<i>Ulmus americana</i>	AE	972	40
Slippery elm	<i>Ulmus rubra</i>	RL	975	18
Other hardwoods	(n/a)	OH	—	19
Total				1024

^a See Alpha code and FIA code descriptions in the FVS southern variant (Keyser 2008).

generally required to transition the existing stand to a balanced uneven-aged condition where yields stabilized. NPV was calculated for the unbalanced condition (initial cutting cycles), and LEV was estimated for the balanced condition, which was assumed to be maintained in perpetuity. In each scenario, 5-15 year cutting cycles were simulated and the optimal cutting cycle was identified based on the highest cumulative NPV produced by both initial NPV during the transition and LEV of the balanced uneven-aged condition.

Economic Analysis

Volume outputs in SN were provided as merchantable stem cubic feet of pulpwood and merchantable stem cubic feet of sawtimber (Keyser 2008). Assumed product prices were based on average hardwood timber prices for region over the period 2003-2014; pulpwood was valued at \$8.43 per ton, non-oak sawtimber at \$24.75 per

ton, and oak sawtimber at \$34.12 per ton (Timber Mart-South, 2004-2013). As SN provided volume outputs in cubic feet, it was necessary to convert to tonnage units for production valuation. Pulpwood volumes in cubic feet per acre were converted to tons/acre based on one cubic foot of pulpwood equal to 0.032 tons, and sawtimber volumes in cubic feet were converted to tons/acre based on one cubic foot sawtimber equal to 0.036 tons (Timber Mart-South 2004-2013). Cumulative NPV was calculated using SN simulated volumes valued using the above estimated product prices. Cumulative NPV for both even- and uneven-aged management scenarios was calculated, assuming a 3-percent discount rate. The difference in NPVs was presented to show the tradeoff between even- and uneven-aged management. Furthermore, Equivalent Annual Annuity (EAA) was calculated (see Nepal and others 2016) to show the annualized dollar value tradeoff between even- and uneven-aged management.

RESULTS

In the even-aged scenario, the existing stand was managed to maximize NPV by following the decisionmaking criteria for managing bottomland hardwood stands (Goelz and Meadows 1997). The maximum NPV was obtained after 40 years of management, which included three thinning treatments: first thinning at time 0 (when management began), a second thinning after 18 years, and a third thinning after 34 years (table 2). During this period the stand produced 5,758.27 cubic feet total volume which included 475.88, 4,307.29 and 975.10 cubic feet for pulpwood, oak sawtimber, and non-oak sawtimber, respectively (table 2). After harvesting the existing stand, the next (second) rotation was managed for 66 years, which produced the optimal LEV. The second rotation produced 6,710.41 cubic feet volume which is the summation from three thinning at ages 32, 46, and 58 after harvesting the existing stand (72, 86, and 98 years from present, respectively) and a final harvest at age 66 (106 years from present). Annual equivalent volume during this second rotation period was 101.68 cubic feet per year per acre, which included 30.85 and 70.83 cubic feet per year per acre each of pulpwood and oak sawtimber, respectively (table 2).

In uneven-aged scenarios, steady periodic revenue was achieved after several cutting cycles depending on the cutting cycle length and selected scenarios. Starting point of steady periodic revenue divided uneven-aged management into two parts: unbalanced (before the steady stage) and balanced (after the steady stage) (table 3). The initial few cutting cycles had more removal because we were

targeting the lower basal area in the uneven-aged management (68 square feet per acre). Unbalanced period in uneven-aged management scenarios had higher proportion of pulpwood compare to the balanced uneven-aged especially with the shorter optimal cutting cycle length (table 3).

For the second rotation onwards, the even-aged scenarios produced relatively higher volume as compared to the balanced uneven-aged scenarios (fig. 1). A balanced uneven-aged condition was identified whenever revenue from each cycle approached a consistent amount. This transition period was needed to achieve a balanced condition in uneven-aged management varied in each scenario. In the balanced condition, annual equivalent volume increased with the q-factor, which implies that if other conditions remain the same that annual volume in the balanced condition increased as the q-factor increased (table 3). Specifically, the annual pulpwood production in the even-aged scenario (second rotation) was higher as compared to the balanced uneven-aged scenarios (fig. 1). Similarly, oak sawtimber production in the even-aged scenario was higher; however, overall sawtimber volume was generally higher in the balanced uneven-aged condition (table 3).

For even-aged management, the optimal cumulative NPV was \$2,499.16 per acre. The optimal even-aged NPV was higher than the optimal uneven-aged cumulative NPV, except for two scenarios (table 4). Stands with 1.5 q-factor and 30 and 38 inches of maximum residual DBH with medium regeneration produced higher cumulative NPVs in uneven-aged management. Overall, medium regeneration

Table 2—Even-aged management intermediate treatments and volume production for both existing stand and the subsequent series of identical rotations managed in perpetuity

Stand ID	Existing stand volume (cubic feet per acre)				Age	Second (perpetual) rotation volume (cubic feet per acre)				
	Pulpwood	Oak	Non-oak	Total		Pulpwood	Oak	Non-oak	Total	Age
Thin I	29.53	0	0	29.53	0	773.95	0	0	773.95	72
Thin II	446.35	0	529.83	976.18	18	1262.0	0	0	1262.0	86
Thin III	0	623.69	445.27	1068.96	34	0	1008.61	0	1008.61	98
Harvest	0	3683.60	0	3683.6	40	0	3665.85	0	3665.85	106
Total	475.88	4307.29	975.10	5758.27	–	2035.95	4674.46	0	6710.41	–
Annual equivalent volume for second rotation (66 years)						30.85	70.83	0	101.68	–

Table 3—Volumes produced by the uneven-aged management scenarios

Scenarios	Volume harvested before balance (cubic feet per acre)				Balanced stage age (years)	Annual equivalent volume after stand balanced (cubic feet per acre/year)				Optimal cutting cycle	
	Sawtimber		Sawtimber			Sawtimber		Sawtimber			
	Pulpwood	Oak	Non-oak	Total volume		Pulpwood	Oak	Non-oak	Total volume		
1.3_24_low	604.55	248.38	263.43	1116.36	14	4.12	37.55	37.77	75.32	79.44	7
1.4_24_low	1043.19	2282.08	1328.27	4653.54	54	7.39	34.71	46.21	80.92	88.31	9
1.5_24_low	1029.88	2617.42	2071.21	5718.51	60	10.4	19.21	59.6	78.81	89.21	12
1.3_30_low	801.56	820.64	562.33	2184.53	28	3.24	38.39	36.06	74.45	77.7	7
1.4_30_low	511.62	88.13	130.85	730.6	7	9.63	35.13	37.33	72.46	82.1	7
1.5_30_low	554.09	143.44	191.12	888.65	8	13.81	31.62	43.74	75.36	89.17	8
1.3_38_low	802.23	818.94	558.32	2179.49	35	2.98	35.91	28.86	64.77	67.75	7
1.4_38_low	539.2	120.6	153.44	813.24	8	7.97	33.66	35.99	69.65	77.63	8
1.5_38_low	554.09	143.55	190.91	888.55	8	14.56	31.65	37.18	68.83	83.39	8
1.3_24_medium	812.68	1297.75	734.41	2844.84	36	1.57	53.06	33.53	86.59	88.15	9
1.4_24_medium	1175.71	2626.61	1527.07	5329.39	63	6.72	48.99	37.34	86.33	93.05	9
1.5_24_medium	1109.52	3213.61	3801.84	8124.97	84	11.56	25.82	55.71	81.53	93.09	14
1.3_30_medium	965.19	2357.58	3217.46	6540.23	88	2.91	37.4	32.6	70	72.92	8
1.4_30_medium	1007.65	1956.04	3011.9	5975.59	72	7.67	34.36	37.35	71.71	79.38	8
1.5_30_medium	1751.01	2830.7	4819.85	9401.56	108	10.39	45.08	35.37	80.45	90.45	9
1.3_38_medium	1131.98	2242.53	1214.51	4589.02	63	0.55	41.13	26.58	67.71	68.26	7
1.4_38_medium	699.44	336.56	304.88	1340.88	16	8.55	42.07	30.58	72.65	81.2	8
1.5_38_medium	1092.79	1596.77	2860.97	5550.53	64	12.91	32.81	38.29	71.1	84.01	8
1.3_24_high	1215.65	3844.03	4858.27	9917.95	132	2.63	10.54	68.83	79.37	82.01	12
1.4_24_high	830.45	2939.04	2585.04	6354.53	77	7.88	6.63	58.95	65.58	72.97	11
1.5_24_high	338.76	349.52	308.81	997.09	15	11.87	36.57	51.54	88.11	99.98	15
1.3_30_high	1039.27	3578.68	2538.51	7156.46	90	3.3	31.63	56.88	88.51	91.81	15
1.4_30_high	866.42	3512.76	4945.31	9324.49	121	7.67	6.79	54.73	61.52	69.19	11
1.5_30_high	618.36	193.08	223.76	1035.2	10	5.97	23.93	45.69	69.62	75.6	10
1.3_38_high	785.22	2997.37	2064.45	5847.04	78	2.08	13.69	54.33	68.02	70.09	13
1.4_38_high	748.72	2193.41	1410.89	4353.02	52	4.96	18.68	56.8	75.48	80.44	13
1.5_38_high	841.59	2280.58	1533.76	4655.93	52	9	22.37	57.05	79.42	88.42	13

Uneven-aged management scenarios in the first row of table represent different combinations of q-factor, maximum residual DBH (inches), and regeneration density. Low, medium, and high regeneration density denotes 10, 20, and 100 percent of average regeneration estimated in table 1 per year per cutting cycle, respectively.

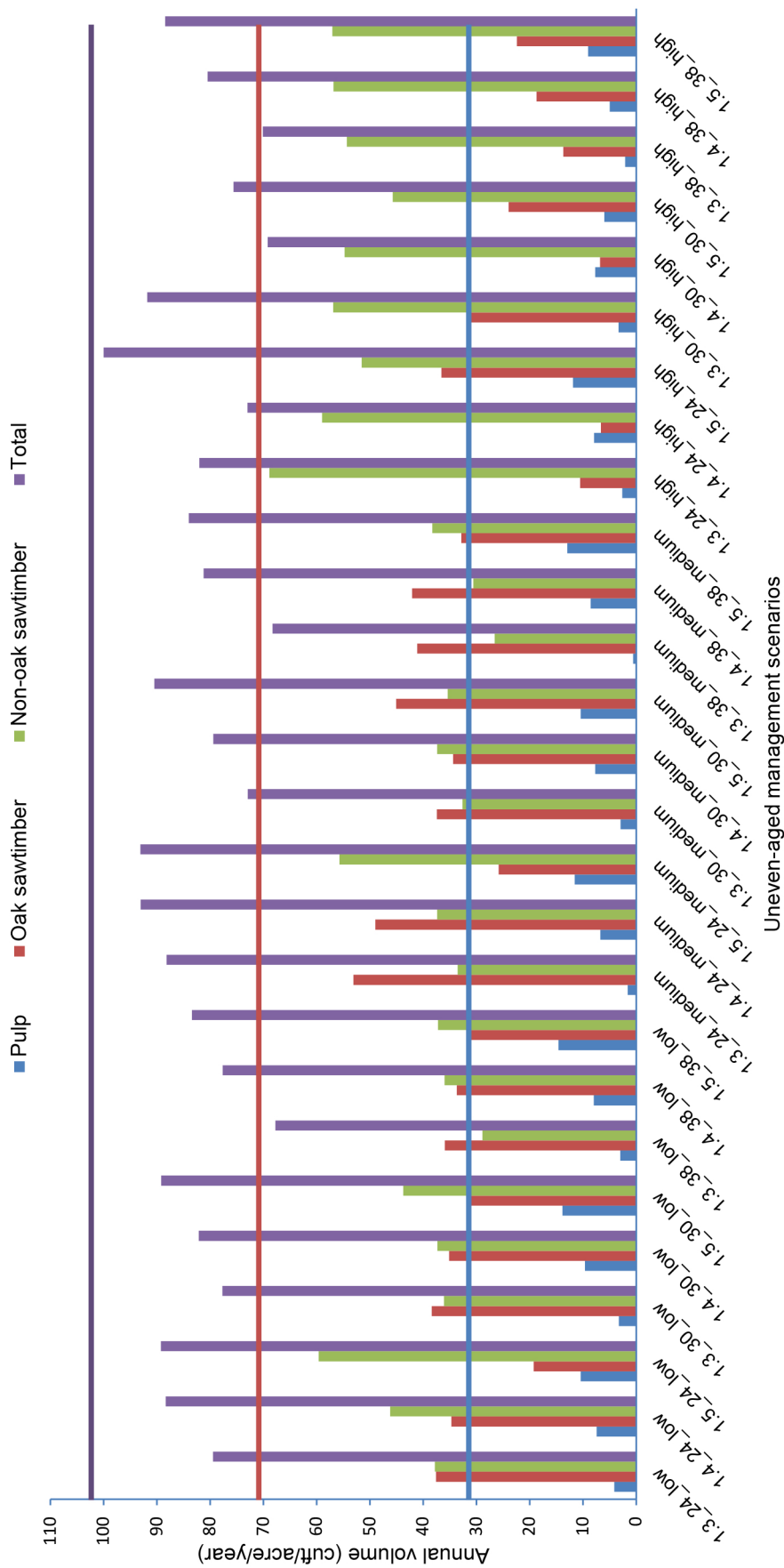


Figure 1—Annual volume produced by even- and uneven-aged management in a balanced condition. Vertical bar graph indicates annual volume produced by each uneven-aged scenario in balanced condition while the horizontal lines of corresponding color indicate annual volume produced with even-aged management at second rotation. The x-axis represents 27 uneven-aged management scenarios indicating q-factor, maximum residual DBH, and regeneration assumption (low, medium, and high regeneration represents 10, 20, and 100 percent of the average regeneration calculated in table 1) For example; 1.3_24_low indicates a 1.3 q-factor, a 24 maximum residual DBH, and a low regeneration assumption.

Table 4—Estimated cumulative Net Present Value (NPV) for uneven-aged management scenarios and tradeoff with even-aged management in terms of dollar value and Equivalent Annual Annuity (EAA)

Maximum residual DBH	Q-factor	Cumulative NPV/uneven-aged			Tradeoff between even & uneven-aged			EAA (tradeoff/year/acre)		
		Regeneration density			Regeneration density			Regeneration density		
		Low	Medium	High	Low	Medium	High	Low	Medium	High
24 inches	1.3	\$2,149.94	\$2,226.74	\$2,044.76	\$349.22	\$272.42	\$454.40	\$10.48	\$8.17	\$13.63
	1.4	\$2,253.99	\$2,283.41	\$2,113.84	\$245.17	\$215.75	\$385.32	\$7.35	\$6.47	\$11.56
	1.5	\$2,371.30	\$2,374.74	\$2,203.64	\$127.86	\$124.42	\$295.52	\$3.84	\$3.73	\$8.87
30 inches	1.3	\$2,089.09	\$2,245.79	\$1,896.16	\$410.07	\$253.37	\$603.00	\$12.30	\$7.60	\$18.09
	1.4	\$2,222.17	\$2,451.28	\$2,021.63	\$276.99	\$47.88	\$477.53	\$8.31	\$1.44	\$14.33
	1.5	\$2,365.99	\$2,610.88	\$2,047.27	\$133.17	\$(111.72)	\$451.89	\$3.99	\$(3.35)	\$13.56
38 inches	1.3	\$1,959.14	\$1,982.18	\$1,905.91	\$540.02	\$516.98	\$593.25	\$16.20	\$15.51	\$17.80
	1.4	\$2,161.14	\$2,165.51	\$2,092.67	\$338.02	\$333.65	\$406.49	\$10.14	\$10.01	\$12.19
	1.5	\$2,231.12	\$2,569.70	\$2,259.20	\$268.04	\$(70.54)	\$239.96	\$8.04	\$(2.12)	\$7.20

Low, medium, and high regeneration represents 10, 20 and 100 percent of average regeneration density estimated in table 1 per acre per cutting, respectively. Tradeoff in parentheses represent uneven-aged management outperformed even-aged.

scenarios produced higher NPVs in uneven-aged which means a smaller tradeoff between even- and uneven-aged management for these scenarios. Tradeoffs decreased with increased q-factors, and as the maximum diameter target was reduced (except for the highest q-factor) (table 4). Among the 27 uneven-aged management scenarios, 25 scenarios produced higher EAA in even-aged management with the highest difference being \$18.09 per acre per year (fig. 2).

DISCUSSION AND CONCLUSIONS

Annual equivalent volume of pulpwood, oak sawtimber, and non-oak sawtimber was estimated and compared for both the even-aged (second rotation) and balanced uneven-aged management scenarios. Volume produced by initial cutting cycles in uneven-aged was not comparable to even-aged because of a variation in the time period; however, volume produced by the second rotation in even-aged and stable cutting cycles in uneven-aged was comparable in terms of annual equivalent production. Even-aged management produced higher total yield over an equivalent time period as compared to the balanced uneven-aged condition. While non-oak sawtimber volume was higher in the balanced uneven-aged scenarios, higher annual volumes of oak sawtimber and pulpwood resulted in higher total annual volumes in the even-aged management scenario. Higher volumes for even-aged management are the primary reason for the higher cumulative revenues as compared to uneven-aged management.

The greater production of oak sawtimber in the even-aged management scenario was a function of the silvicultural constraints in the scenarios and competition dynamics in the model. Since we preferentially favored oak to retain during thinnings in the even-aged management scenario, it was obvious that more oak sawtimber would be produced at final harvest. Although oak sawtimber volume was higher in the even-aged management scenario, the combined sawtimber volume was higher in most of the uneven-aged management scenarios. Since the oak sawtimber product price was higher than non-oak sawtimber, the overall cumulative NPV was higher for even-aged management. In the second rotation, the first two thinnings produced pulpwood because

these treatments were applied as thinnings from below. Sawtimber was obtained only after the third thinning and at final harvest. In contrast, the uneven-aged management scenarios produced little pulpwood and a greater proportion of sawtimber at each cutting cycle harvest so as to maintain the specified q-factor. As a consequence, overall pulpwood production was lower in uneven-aged scenarios as compared to the even-aged management scenario. In addition, no species preference was applied in the uneven-aged management scenarios, thus stand development was a function of shade tolerances of the different species, as parameterized within SN. As oaks are less shade-tolerant than many of their associated bottomland species, they had a lower probability of recruiting to the forest canopy and becoming sawtimber trees in the uneven-aged stands conditions where a residual canopy is continuously maintained.

Stand structural parameters greatly affected the economic tradeoffs between approaches. As the q-factor increased and other conditions remained the same, the total volume production increased and thus cumulative NPVs increased. The target residual basal area of 68 square feet per acre required the cutting of more small size trees to maintain a lower basal area under the higher q-factor structure. As a consequence, annual production of pulpwood as well as the total volume production was higher with a q-factor of 1.5 as compared to q-factors of 1.3 and 1.4 (table 3). NPVs produced by the initial transitional cutting cycles before the stand reached a balanced condition had a substantial impact on the cumulative NPV, especially if the stand required more time to reach the balanced condition. In these cases, the early revenues produced by the initial cutting cycles were substantial, and the LEV from the balanced condition had less impact on cumulative NPVs. In addition, as the maximum diameter target was reduced, from 38 to 24 inches, the economic differences between the approaches narrowed, at least at the lower q-factors. This suggests that, at least for q-factors of 1.3 and 1.4, the tradeoffs can be minimized by using a lower (24-inch) maximum diameter. Medium regeneration density scenarios produced better cumulative NPVs as compared to low and high density. In fact, the medium regeneration produced the highest NPVs among the regeneration density if compared only

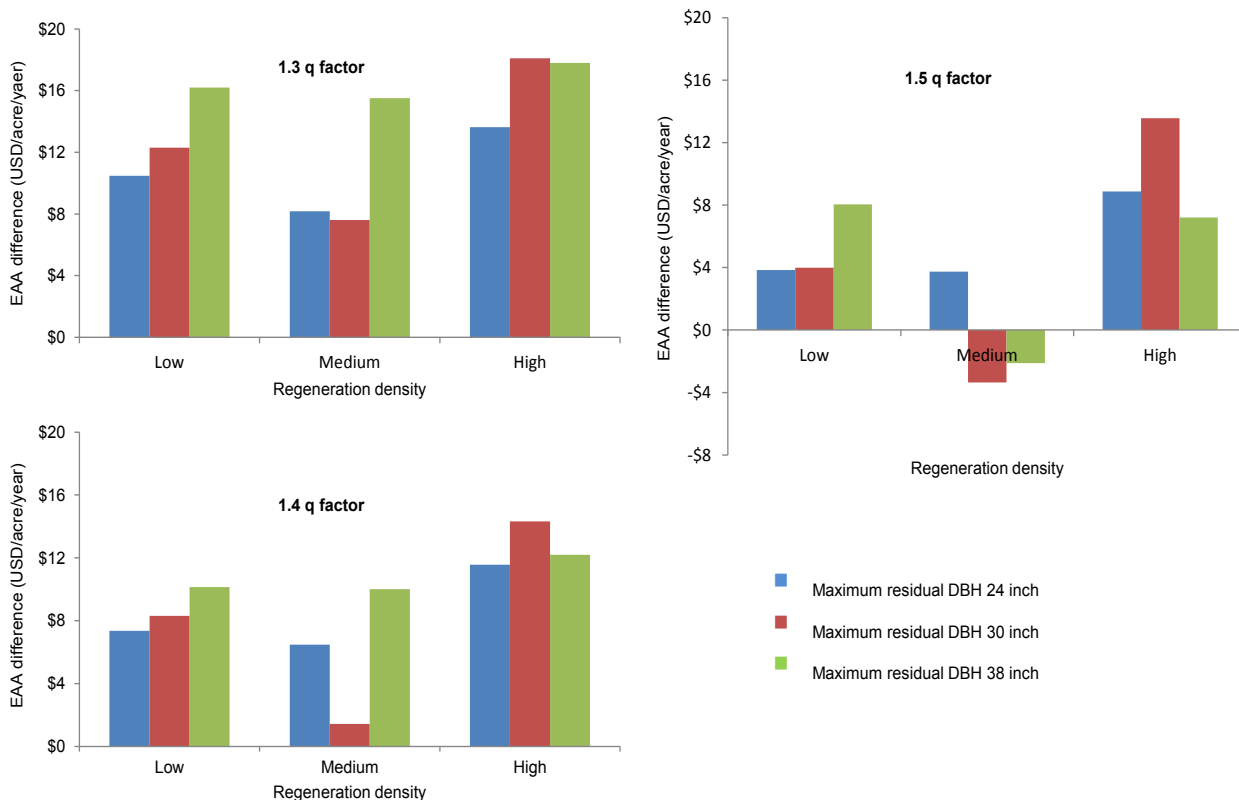


Figure 2—Equivalent Annual Annuity (EAA) difference in terms of U.S. Dollars (USD) per acre per year between even- and uneven-aged management. Positive EAA indicates that even-aged management performs better while negative EAA values indicate that uneven-aged management scenarios perform better. In each plot, the x-axis represents regeneration density, each bar represents maximum residual DBH, and the y-axis represents EAA difference.

with regeneration density with all other variables the same (table 4). It also indicates that higher regeneration densities did not guarantee the optimal NPV, presumably due to higher levels of competition and higher mortality.

Out of the 27 hypothetical uneven-aged management scenarios, only two produced higher cumulative NPVs compared to the even-aged. Those two scenarios were from the higher q-factor (i.e., 1.5) scenarios, with medium regeneration density and higher residual maximum DBH. In all other scenarios, even-aged management outperformed, but the magnitudes were different and depended on the selected variable in each scenario. Overall, NPVs in the uneven-aged management scenarios were sensitive, to at least some degree, to all three parameters: q-factor, maximum residual DBH, and regeneration density.

In summary, even-aged management was economically superior in the majority of the cases. It should be noted, however, that this analysis was based on timber valuation only, and with a specific set of stand structural assumptions and silvicultural methods. Future research should consider other silvicultural approaches for achieving structural variability, as well as address nontimber values and management costs.

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INDEX OF AUTHORS

Adams, Joshua	172, 182	Kralicek, Karin M.	74
Ager, Alan A.	129	Kuehne, Christian	14
Arciniega, James	154	Larsen, David R.	72, 94
Asherin, Lance A.	89	Linn, Rod	37
Azpeleta, Alicia	126	Loreno, Sara	40
Bagdon, Benjamin	126	Lu, Hsien-chih Bryan	64
Bailey, John	129	Marston, Jonathan	164
Battaglia, Michael A.	89	Martin, Fred	64
Bell, Conor K.	40	McDaniel, Virginia L.	98
Berretti, Roberta	18	Mcmahan, Andrew J.	149
Beukema, Sarah J.	34	McTague, John Paul	14
Burton, Jesse A.	98	Mell, Ruddy	37
Cohn, Greg	37	Monahan, William B.	149
Crookston, Nicholas	4, 10	Nagel, Linda M.	2
Day, Michelle A.	129	Negro, Matteo	18
Deo, Ram K.	52	Nepal, Sushil	126
Dey, Daniel C.	72	Nowak, John T.	140
Dickinson, Yvette L.	89	Oppenheimer, Mike	14
Drees, Dan G.	98	Parsons, Russ	37
Ex, Seth A.	110, 114	Pimont, Francois	37
Falkowski, Michael J.	52	Rathbun, Leah C.	74
Forder, Melissa M.	98	Ray, David	78
Fried, Jeremy S.	40	Rebain, Stephanie	110
Froese, Robert E.	52	Regazzoni, Elena	18
Ghilardi, Casey R.	94	Robinson, Donald C.E.	34
Glasby, Macklin	87	Rodrigue, Jason A.	140
Haberland, Christopher	164	Roske, Molly	2
Havis, Robert N.	24	Ruffinatto, Flavio	18
He, Hong S.	94	Russell, Matthew B.	52, 87
Hennigar, Chris	10	Sánchez Meador, Andrew	74, 126
Hoffman, Chad M.	37, 114	Seli, Robert	129
Holley, Gordon	172, 182	Seymour, Robert	78
Hoover, Coeli M.	60	Shettles, Michael A.	4, 57
Houtman, Rachel M.	129	Smith, Alistair M.S.	114
Jain, Theresa B.	40	Smith, Frederick (Skip)	110
Johnson, Ralph	64	Smith, James E.	60
Jolly, W. Matt	37	Smith-Mateja, Erin	8, 57
Kabrick, John M.	72, 94	Stoddard, Michael	126
Keefe, Robert F.	40	Tinkham, Wade T.	2, 114
Kershaw, John	10	Vacchiano, Giorgio	18
Keyser, Chad E.	140	VanderSchaaf, Curtis L.	82, 172, 182
Keyser, Tara L.	98, 110	Vickers, Lance A.	72
Klein, Robert N.	98	Weiskittel, Aaron	10, 14
Knapp, Benjamin O.	72, 94	Wells, Lucas	37

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Keywords: Economics, forest health, forest management, forest prediction models, growth and yield modeling, landscape dynamics, model validation and evaluation, regeneration, vegetation dynamics, wildlife habitat.

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