REVIEW PAPER

# Scaling Issues and Constraints in Modelling of Forest Ecosystems: a Review with Special Focus on User Needs

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*Abstract*

Forest Ecosystem models are diagnostic tools to assess and understand ecosystem processes. Conflicting interests such as simplicity, observability and biological realism must be addressed to ensure a well balanced modelling approach. Because field observations are usually only available for short time periods or for a limited number of locations, models are important to extrapolate in space and time. The key to a successful modelling approach relies on finding the appropriate scale but is often limited by the availability of input data. In the practice of forest modelling, it is often necessary to produce meaningful outputs on the basis of rather weak information. In this paper we discuss spatial and temporal scaling issues within empirical modelling. Following the case studies the paper demonstrates how models address cross-scaling problems as they are relevant for the required data as well as the decision making and implementation process of specific end user needs.

**Key words:** forest modelling, scaling, end user

## Introduction

Forest ecosystems and their management play an important role in achieving sustainability goals since they are important for economic, social and cultural reasons. Field observations for any forest related phenomena are usually available for short time periods or for a limited number of geographical locations, therefore models are an important tool to extrapolate this empirical knowledge in space and time (Pacala et al. 1996). While in the past forest management practice was considered to be the main driver of ecosystem change, recent research suggests that climate change and the associated risks will be increasingly important in the sustainable development of forest ecosystems (Eastaugh et al. 2011, Hickler et al. 2012). Forest management also affects the susceptibility and resilience of forests to potential climate change impacts. In addition, our understanding of ecosystem processes has increased and a large number of different data sets and monitoring programs are available to assess forest ecosystem changes.

With the development of computer technology and the improvement in our ecological understanding, a wide variety of models have been developed. Scientists define a model as a simplified representation of a more complex reality (Gadow 1996, Shugart 1998). Models will always contain some level of abstraction by creating a system, which is less complex than the reality, but are still able to describe the pattern and behaviour of the real world (or at least the parts of it that the modeller is interested in). Models have been developed in many different forms, including conceptual and graphical models, statistical models and computer simulation tools (Kimmins 1987).

Most forest models are developed for specific purposes. They differ with regard to *generality*, the applicability of the concept to a range of instances, and *precision*, the degree of exactness in the predictions (Sharpe 1990, Gadow 1996). High precision is often achieved at the cost of low generality and the choice of model usually represents a compromise.

Each model needs a ‘unit’, the particular quantifiable aspect of the system that is being studied or predicted. While this is clear for traditional growth and yield production models (e.g. standing volume, increment etc.), it may be difficult to define within biogeochemical/mechanistic modelling, as these models must integrate various ecosystem processes at different spatial and temporal *scales*. ‘Scale’ in this context refers to the “spatial and temporal dimensions of a pattern or process” (Cumming et al. 2006) and has been described as “the central problem in ecology” (Levin 1992).

When using ecosystem models, it is important to understand how a model responds to both normal undisturbed conditions and also to assumed disturbances. Scenario simulations are important in sensitivity analyses and are a major step in evaluating and testing the reliability and realism of expected model output, particularly if adequate test data sets are not available.

Model development is often based on a system analytical approach (Figure 1). First, the problem must be clearly formulated. The objectives should be defined observing the three criteria as simplicity, observability and biological realism. In this step, the framework of the model functioning (its application and expected output) is defined (Hasenauer et al. 2000). Data collection is expensive and time consuming, and, therefore, many of our modelling efforts are limited by data constraints. This sets limits to the opportunities for optimum scaling, both spatially and temporally. If a model is to be applied on a larger scale or to a longer time period than that to which it was calibrated, then the data structure used for calibration and model initialisation can have a major impact on the reliability of the expected output (Sims et al. 2009).

Every modelling project must include a model validation and/or evaluation. Any model should be based on the latest available knowledge to ensure reliability; therefore, the validation should be a continuous process. To avoid ‘data mining’ and spurious results, data used for assessing the expected error range of key output variables must be different from that used for calibration (Green 2002). One important objective in modelling is to mimic nature as well as possible (Kangur et al. 2005), maintaining the biological realism of the output. This may require procedures or experiences which are not always rigorously tested by typical statistical validation procedures.

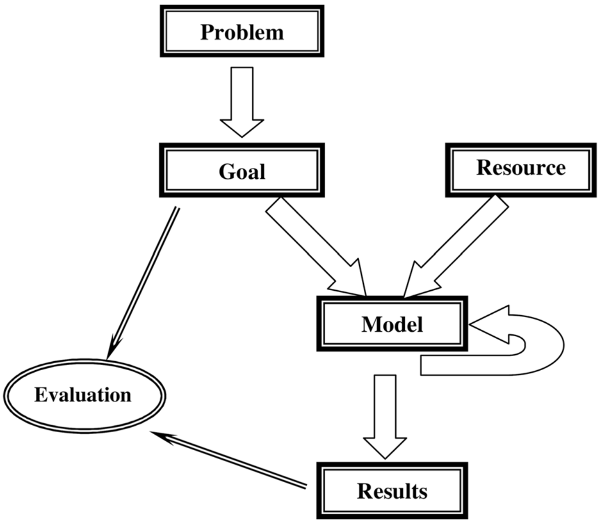


**Figure 1.** System analytical approach to design ecosystem models (Hasenauer et al. 2000)

Although all models seek to faithfully reflect the relevant states or processes in a system, there are important constraints that can limit their effectiveness. An important constraint in today’s modelling efforts is the fact that in some areas of research our fundamental scientific knowledge is limited. In such situations models help to detect research needs and to design field studies. Simple input/output approaches are suitable to be used until a comprehensive understanding of missing research information becomes available. Even where processes are reasonably well understood, it is often difficult to accurately monitor them or describe and quantify them in ways meaningful for mathematical modelling. This problem is particularly acute when our process understanding is at an entirely different scale to the scale of the outputs required by the end user of the model.

Although issues of scale have been extensively studied in ecology (i.e. Wiens 1989, Hobbs 2003, Chave 2013), the issue has not gained much attention from the users of forest models. The aim of this review paper is to present a discussion of different aspects and constraints of scaling in forest ecosystem modelling. Our goal is to bring an appreciation of these issues to the end users of forest model outputs.

### *Theory and concepts*

#### *Scaling in forest ecosystem modelling*

Ecosystem processes act at many different levels (Figure 2), and this will require different procedures for data collection and integration. Some processes may be on a tree or leaf level (such as photosynthesis), others may act over a regional or even global scale (e.g. expected change in climate). Similarly with temporal scales, some processes operate within seconds or less (gas exchange within the stomata) while others take years or centuries to have an appreciable impact on an ecosystem (soil formation). Any process may in theory be reduced to the level of atomic and inter-atomic forces, but this is rarely meaningful in studies of ecosystem dynamics. Similarly, processes may be studied at the global scale, but for many instances this does not provide useful answers. If we are interested in the resistance of variously-structured forests to windthrow, for example, neither a study of the interactions of the air molecules with the molecules in a tree nor an examination of solar output over the past thousand years are likely to be useful. These differences in scaling are critical concerns when we wish to combine different processes. Formal conceptual consideration of the three major modelling criteria (i) simplicity, (ii) observability and (iii) biological realism is essential to ensure in a well balanced model (Green 2002). The term ‘well balanced’ refers to the fact that it makes no sense to describe one model driver very carefully and in great detail while other important drivers are less rigorously implemented or even kept constant. Therefore, ignoring biological realism may lead to poorly defined or misleading modelling results. From a statistical perspective random variation may be removed and such models pretend to produce accuracies in their predictions which do not truly exist.

#### *Modelling concepts based on user needs and requirements*

At the beginning of any model development or modelling exercise there is a practical or theoretical problem which has to be solved or needs to be structured and understood. Therefore, depending on the topic of interest and the background of the model developer and/or end user, a variety of models have been developed. From a conceptual point of view we may distinguish between the following forest ecosystem modelling concepts (Figure 3, see also (Hasenauer et al. 2000)).

Growth and yield models are the oldest and traditional approach to forest planning. Such models are designed to forecast forest growth to ensure sustainable timber harvesting. Based on forest inventories and yield tables (Weise 1880), the mean stand development over time – one rotation period – is described. It is important here to recall that yield tables were developed for even-aged, monospecious stands. For management purposes in such stands, the yield tables of the late 20th century were quite adequate. If all trees in a stand are the same, then there seems a little purpose in modelling individual trees as separate entities. When end users started to demand models, that could be applied in the uneven-aged, multi species stands, then the ‘yield table’ approach became inadequate and the individual tree models became more popular (see Nagel 1999). The main advantage of individual tree models is that they are independent of any specific

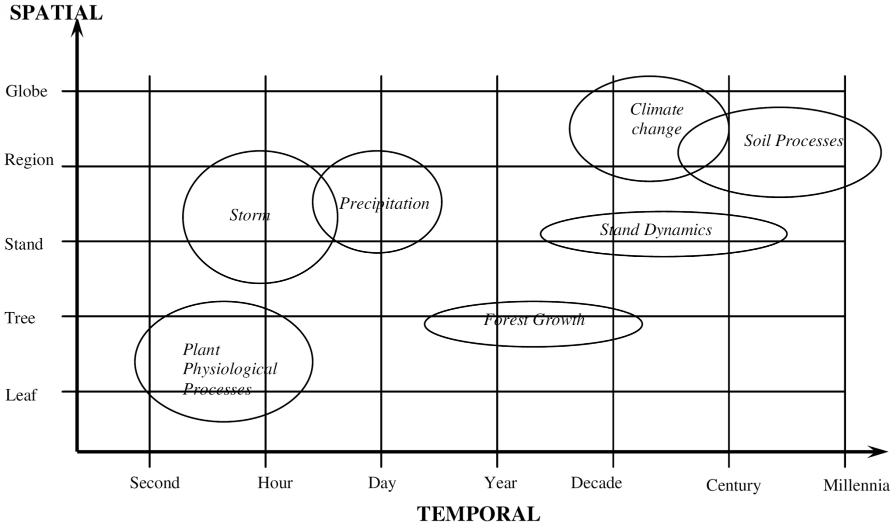


**Figure 2.** Modelling of ecological processes operates at various temporal and spatial scales

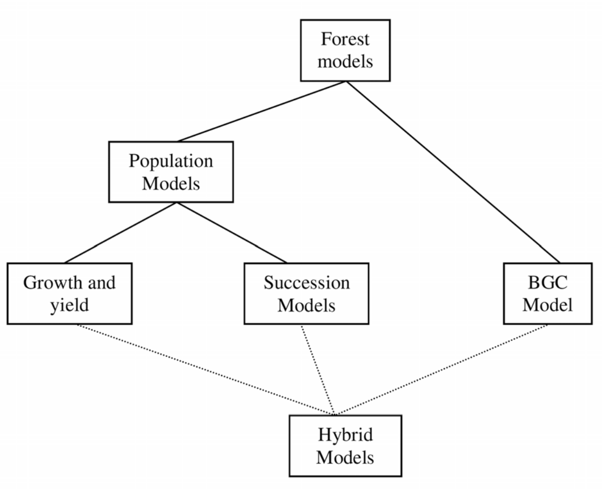


**Figure 3.** The three most common forest ecosystem modelling concepts

mixture, age or stand treatment, because the growth and mortality of each tree within a stand is described. Therefore, they can be applied in situations where traditional yield tables cannot be used.

Succession, Gap or Patch models describe the reproduction, growth and mortality of trees (Botkin 1993). Old trees die resulting in forest gaps or patches, where new trees will grow (Shugart 1998). The first and most famous succession model was JABOWA (Botkin et al. 1972), developed for describing the stand dynamics of the Hubbard Brook experimental forest in the Northeastern US. Since then, numerous applications have been developed (i.e. Shugart 1998, Pacala et al. 1996, Lexer and Hönninger 2001, Bugmann 2001).

Although tree growth is simulated, the main goal of succession models is to describe vegetation patterns over time. The stand dynamics as it has evolved during centuries was and still is one of the main features. The regeneration, growth and death of individual trees and the interaction between different tree species are the key features. Management impacts are usually not assessed and the interactions between the different main driving forces of growth (energy, temperature, water, nutrients) are the main research areas to enhance the functioning of this model type. Today, such models are used for assessing the potential vegetation patterns and changes in the vegetation distribution under expected climate change.

Mechanistic models describe the circulation, transformation and accumulation of energy within and through a forest ecosystem. Their main goal is to describe the interactions between plants and the surrounding environment; therefore, such models are often called process or biogeochemical-mechanistic models (BGC-Models). In contrast to tree population models such as growth and yield or succession models, BGC models incorporate a mechanistic description of the interaction between plants and the surrounding environmental conditions (Waring and Running 1998). Consequently, they are designed to be responsive to changes in environmental conditions. BGC models operate on a monthly to daily time resolution to simulate the cycles of carbon, water and nitrogen for generalized biome types (Thornton 1998) or species (Pietsch and Hasenauer 2002). Leaf area index (LAI, $m^2$ leaf area per $m^2$ ground area) controls the canopy radiation absorption, water interception, photosynthesis, and litter inputs to the detritus pools. Further details related to concepts in ecosystem modelling may be found in (Hasenauer 2005).

Different information demands may require the different temporal and spatial scaling. Scaling is part of the formal problem and addresses the fact that information becomes an emerging property at a certain temporal and spatial scale. If the level of “scaling” is ill-defined, then interesting effects might be masked by the level of detail (e.g. the noise hides the signal) or the generalisation is too coarse which levels off the modelling effects of interest. A ‘simple’ problem needs to be defined in terms of scale, at what level the information should be generated and produced. In more complex models, this may not be straightforward as sub-routines in the model may work at different scales and, therefore, impact the output. Furthermore, there may be a lack of awareness on the part of decision makers (end users) on the importance of scaling issues for making decisions in which ill-structured situations are usually appear, often involving poorly defined questions.

Models are sometimes developed to meet specific user demands for information or decision guidance. These models are commonly based on other models that were designed for research purposes understanding processes. While models can often be adapted from research tools to management or policy tools, the scientific integrity of the model should be maintained. This requires a careful assessment of how temporal and spatial scaling resolution may differ in the new application. An inappropriate choice of scale may produce deficient studies in two ways: through a masking effect of processes operating at different scales or through a loss of discriminatory power. If full information on all processes was known at a sub-molecular level, then a complete system could be perfectly modelled, but this is impossible in practice.

Information and understanding of processes is often not available at the temporal or spatial scale required to provide meaningful direct outputs. Furthermore, there is often a mismatch between the temporal or spatial scale of the known data, the level of understanding of the processes and relationships, the computational power realistically available, and the desired outputs. Therefore, some means are necessary to convert information from one scale to another, often at several different levels within one modelling exercise. There are different potential reasons for insufficient information, e.g. (i) information was not recorded, (ii) it is impossible to measure information needs, (iii) change of the key drivers along the modelling process. The impacts of the sequence of processes (e.g. planning and implementation, using sub-modules in modelling environments) have an important impact on scaling demands and needs.

Scaling may be required in either (or both) spatial and temporal dimensions. Upscaling denotes a change to a coarser, more generalized representation of the data, either using larger spatial units or longer temporal periods. Downscaling conversely infers information at a finer resolution or shorter time period than the original.

The construction of a forest ecosystem model represents an upscaling process. BIOME-BGC (Running and Hunt 1993) simulates all major essential ecosystem processes and, therefore, is able to track the development of a forest stand under specific and changing environmental conditions. In designing, this model the various plant parts (cell organelles, cells, small tissues and organs) and other elements of the system were generalized into key ecosystem compartments, i.e. sun and shade leaves, live and dead stem, fine and coarse roots, litter and coarse woody debris and soil (described by its texture and depth). The range of fine and coarsely scaled processes were then reduced to the minimum number necessary for representing the key ecosystem fluxes of energy and the most important substances (carbon, water and nitrogen). Carbon uptake through the stomata, photosynthetic fixation and allocation, litterfall and mineralization, precipitation interception, water uptake through roots and loss through leaves and many more other processes are calculated as functions of the meteorological variables, site conditions and species-specific ecophysiological parameters. A forest stand is modelled as homogenous, which means that only the dominant tree species are simulated and conditions do not vary within a stand (defining the output scale of the model). Within these simplifications, fundamental upscaling steps have been included in modelling.

Downscaling derives values at the finer spatial scales or shorter temporal periods from information at the coarser or longer scales. Often this is done by simply assuming that all values at the smaller scale will be equal to those at the large (e.g. soil types and depths are often simplified as being spatially constant at fine scales, based on estimates of the broad-scale mean), although more sophisticated interpolation methods may be used to ‘smooth’ the transitions at the fine scale (Petritsch and Hasenauer 2007). Temporal downscaling may assume that the short scale values change in definable ways, and may, therefore, be inferred from the longer scale data. For example, if temperature over a 24 hour period is assumed to follow a particular sinusoidal pattern, the knowledge of maximum and minimum temperatures at a daily scale may be used to infer hourly values (Chow and Levermore 2007). The accuracy of these values will depend on the accuracy of the assumed pattern.

Models may be constructed using either a ‘top down’ or ‘bottom up’ perspective. The term top down vs. bottom up may be seen as the general approach in addressing ecosystem processes within a modelling environment. Top down models start from a higher level of resolution and zoom in a more detailed problem while a bottom up modelling approach derives information for a given spatial or temporal resolution from a more detailed source. The term “source” used in this context refers either to the temporal or spatial resolution of available data but also to concepts in deriving ecosystem processes.

A typical example of top down data issues are monthly mean climate values which are used to mimic daily weather patterns by applying a weather generator. Information at higher temporal scale (the month) needs to be processed to have daily values (lower temporal scale) available. It is clear that such top down generated daily weather data may have a different error structure versus observed daily values, and this may have a strong impact on the resulting predictions. A typical bottom up data issue is the use of inventory data where a certain minimum number of inventory plots are required to have a reliable assessment of a given forest area depending on the variability of the measurements and the required accuracy of the output. A single inventory point does not represent a value that can be assumed to be ‘representative’ of a broader area, but when sufficient points are aggregated together, they can provide an accurate estimate of the mean for that area.

From a conceptual perspective, a typical example for top down versus bottom up may be the prediction of growth increment within tree population modelling by using either (1) the potential dependent (top down) or (2) potential independent (bottom up) functions (see Hasenauer 2005). Potential dependent increment predictions were first proposed by Newnham (1964) and assume an upper limit or a limited growth. This upper limit or growth potential is species and site specific. Models based on this concept must first define increment (diameter and height) potentials. These potentials are then reduced for each tree within a stand according to the specific competition situation of each tree, and can be derived using crown length and competition indices. This "potential growth" concept has been also implemented within the gap modelling (see Botkin et al. 1972) by defining a species specific potential. Actual growth rates within a gap model are derived using this potential and reducing it to model the actual growth according to varying site conditions and the competitive status of each tree within the stand.

The bottom up example in this context is implemented in models where the growth increments (diameter and height) are derived from the available data and are only based on a set of independent variables. No upper limits or maximum growth increment rates are defined. Site variations as they may affect tree growth are part of the dependent variables. This concepts has been implanted in the Stand Prognosis Model (PROGNOSIS; Stage 1973) and PROGNAUS (the Austrian version of PROGNOSIS constructed using data from the Austrian National Forest Inventory; Monserud and Sterba 1996). Similar conceptual approaches are found within biogeochemical mechanistic modelling approaches (Running and Hunt 1993).

The important end user related issues here are what type of data are required and what are the advantages and disadvantages in using a top down versus bottom up conceptual approach to tackle an end user problem. Ideally, a model should be as simple as is possible to meet its required purpose. In practice, this has usually led to a preference among end users for top-down empirical models. More recently, however, end users have been asking more sophisticated questions, often relating to aspects of systems outside previously observed ranges (such as the likely response of forests to climate change). As top-down models become more complex and better, understanding of processes allows further simplification of bottom-up models (i.e. Wang et al. 2009); there may perhaps be a 'convergence' of approaches (Figure 4).

The choice of top-down or bottom up models depends largely on what question needs to be answered. Although methods exist for transforming outputs in one form to another (i.e. deriving stand carbon sequestration from diameter measurements), these conversions themselves may introduce errors into the results that would not be present if a more direct model was applied. Figure 5 shows a conceptual view of how three model types may all give outputs in terms of stem carbon, biomass or timber volume, but in less or more direct fashions. An inventory is a statistical model of a forest, which can be upscaled to give a representation of a tree population. Such a population is the direct output of single tree growth models or growth and yield models. Allometrics may be applied to a population to give estimates of timber volume, and then expansion factors used to give outputs in terms of biomass, which in turn can have their carbon fractions estimated. The process model BIOME-BGC, however, has ecosystem net primary production as a direct output, which then requires assumptions to be made as to allocation ratios to derive stem carbon. The carbon fraction and expansion factors may then be applied to estimate biomass and timber volumes. As each of these conversion processes are statistical estimates, errors and biases may be introduced or exacerbated. Even at what would appear to be the simplest point

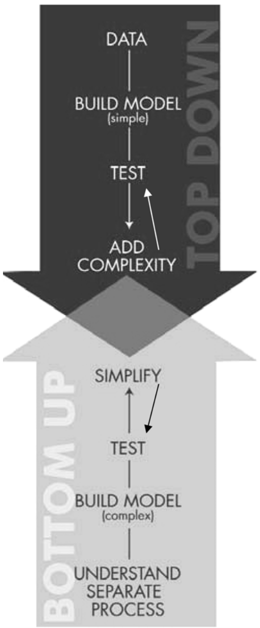


**Figure 4.** Top-down bottom-up convergence (adapted from Catchment Hydrology CRC, 2005)

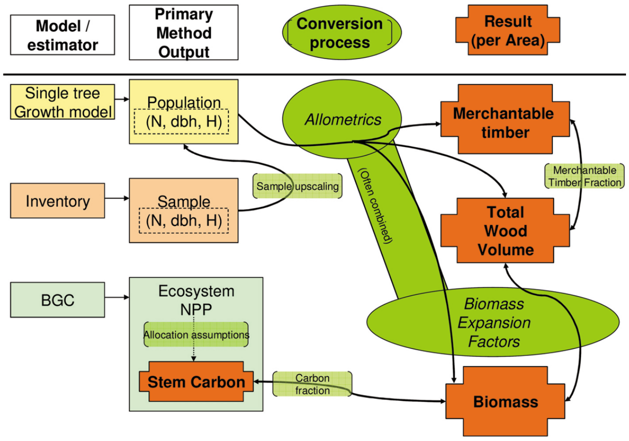


**Figure 5.** Conceptual view comparing outputs from three model approaches, and how each may be used to derive common outputs.

(the upscaling of sample results to a population), biases can be introduced. Roesch and Van Deusen (2010) offer the example of the United States Department of Agriculture Forest Service's Forest Inventory and Analysis Program, where a change in methodology from angle-count to nested fixed-area plots led to marked differences in upscaled diameter distribution output, even though both alternative approaches were theoretically unbiased.

Although in theory all approaches should give the same answers, much depends on the accuracy of the conversion steps. Allometrics, biomass expansion factors and merchantable timber proportions are often highly variable, and generic or average values may not be applicable to the study in hand. While it expected to be clear that data should not be upscaled from a few sample plots to represent a very broad scale mean, it is equally true that broad scale means should not thoughtlessly be applied at the plot scale. In practice, this is often unavoidable due to a lack of precise information at the required scale, but modellers and end users should be aware of the potential for error. An example of this flaw is the conflicting estimates of carbon storage in Australian temperate forests: The IPCC default value of 217 tonnes per hectare derives from the mean global estimates, but Mackey et al. (2008) extrapolate from a single local study area to give an estimate of 640 tonnes/ha. For an end user, neither value should be accepted without a clear understanding of its provenance and limitations.

Top-down modelling approaches use broad-scale input values, generally developed from statistical sampling procedures such as National Forest Inventories. An inventory value for a single site has a little meaning, as this represents just one sample from the population. Bottom-up models, however, assume homogeneity at the scale of their input data (Landsberg et al. 1991), and each site is often assumed to be representative of a larger area. At a broad scale, outputs should be compatible, where sufficient samples are available for statistically valid estimates in the top-down modelling and for adequate differentiation of heterogeneous sites in the bottom-up approach. For single (or few) samples, however, comparisons of results between approaches are likely to be meaningless. Therefore, the point of model convergence (cf. Figure 4) is dependent on the scale of the comparison.

### *Accuracy and precision constraints in model scaling*

As discussed earlier, the key to successful modelling relies on finding the appropriate scale. For modelling, the effects of climate change on forests subnational studies must be made to reflect regional variability (Andreassen et al. 2006, Eastaugh et al. 2011). Model outputs may vary from a reality in terms of the accuracy or precision (or both). The accuracy refers to how close the *mean* model output is to the true value, while precision is a measure of the spread of the output (Figure 6). Poor accuracy (bias) must be avoided in modelling, but may be relatively easy to correct through adjustment of input assumptions. Poor precision, however, suggests either that the system (and/or elements of the model) contain high random variability, or that the sensitivity of the model to particular facets of the problem is inappropriate. This could mean that the model is oversensitive to relatively unimportant changes in the system, or that the model is insufficiently sensitive to (or independent of) important differences.

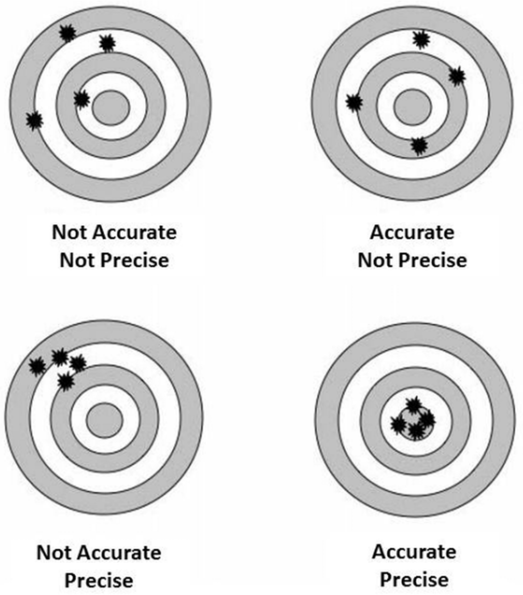


**Figure 6.** Accurancy and precision

Perfect accuracy and precision are unobtainable, as models will always be a simplification of reality. Data availability is often limiting, and in some cases, processes are not fully understood in a physical sense.

Modelling is sometimes called on to fill knowledge gaps (Sims et al. 2009), or to analyse scenarios of events that, for some reason, are not currently easily measurable (most obviously, events in the future). Although modellers understand that such extrapolations will contain an element of error and uncertainty, end users may not. For this reason, the Intergovernmental Panel for Climate Change is careful to label their prognostications on the future as 'projections' rather than 'predictions'. The IPCC (2001) defines a projection as:"... *a potential future evolution of a quantity* *or set of quantities, often computed with the aid of a model. Projections are distinguished from predictions in order to emphasise that projections involve assumptions… that may or may not be realised and are therefore subject to substantial uncertainty*".

However, the innate 'fuzziness' of model outputs is often difficult to quantify and express, especially in a political or management context where the end users seek firm evidence to support or dismiss particular options. Silberstein (2006) points out the risks, that "*… as high level managers appreciate the nice graphics and, possibly, simplistic sets of options* [produced by models]*, it can be easy to lose sight of the limitations of the process that generated them".* Although models may appear to substitute for gathering expensive data or waiting for long periods (perhaps decades) to see experimental results, the adage that "All models are wrong, but some models are useful" should be remembered.

Changing scale may introduce or exacerbate errors and biases, or hide important differences. Eastaugh et al. (2010) compared Austrian temperature data obtained from three sources ('high value' meteorological stations, gridded data at a 10' resolution downscaled from a broad-scale 0.5° European dataset and site-specific DAYMET interpolations) and found substantial differences. The gridded and interpolated data have showed very similar mean behaviour, but the gridded data smoothed out much of the spatial variability that was apparent in the interpolated dataset. The station data have showed a consistently lower national mean temperature than the others due to the stations having a lower mean elevation. Therefore, the choice of appropriate input data for modelling is tied to the scale of the model and of the required output. While downscaled gridded data may be adequate for studies of national-scale temperature trends or long-term hydrological models (for example), mechanistic forest growth models are likely to need climate data that is more specific to the precise locations being studied. Coarse-scale studies may be equally accurate at that scale, but imprecise at finer scales due to a loss of discriminatory power.

### *User prescribed scaling examples*

Evidently, different modelling concepts are in use to meet particular end user needs. Models are often developed in a research situation to understand particular aspects of forest systems, but as Vanclay (2003a) notes, the ultimate aim of many modelling groups is to see their model used for estate planning. Models may also be used as a communication tool, to present common information to a diverse range of stakeholders and enable cooperative policy development (Vanclay 2003b).

User groups may be defined by the purposes for which they require information/modelling:

(i) Understanding the system and models as research tools,

(ii) Management applications,

(iii) Aggregated policy related information tools.

Although researchers may be interested in modelling at any spatial or temporal scale, other model users may have more limited interests. Traditional commercial forest management focuses on one or two rotations of the stand across the extent of the forest estate. Increasing environmental awareness has expanded both the temporal and spatial aspects important for forest management, as in some cases ecosystems are expected to remain essentially unchanged forever ('preserving' old-growth forest), and many environmental pressure groups profess a 'global' approach to environmental management. Aspects and impacts of the forest, that exist spatially 'beyond-stand' such as carbon sequestration or hydrological effects, are of interest to policy makers, while local stakeholders may tend to take a more spatially explicit approach. Successful applied modelling depends on answering the right questions, which in turn depends on informed discussion with end users.

Models may be used alternatively as diagnostic, simulation or forecasting tools, each use having different input requirements, caveats and risks. The application of a model outside its original design purpose should not be attempted without a full understanding of model logic and limitations, as tempting as it may be. This is of particular importance where models are used to support particular ideological positions or policy prescriptions. In an environment where contentious environmental decision-making is based increasingly on complex models, modelling must be as follows:

- completely transparent in all its aspects (data input, processing and output interpretation),
- understandable by all parties, and
- 'mature', in the sense that the model has been repeatedly shown to be useful in comparable contexts.

In the following section, we provide examples to demonstrate end user related scaling issues and why they are relevant within ecosystem modelling. The examples are selected according to their regional importance, and to demonstrate how scale must be considered in a variety of modelling applications.

### *Stakeholder participation*

Forest hydrological modelling is frequently needed, especially regarding conflicts of water use by forest plantations and industry (Srdjevic 1987, Bosch and Gadow 1990) and in areas where catchment management affects the yield and quality of stream water used by industry and households (Murray and Gadow 1991). We present the Nadela river case study here as an example to demonstrate a scenario of motivating several key players to participate in issues related to the operation and long life development of the hydrologic system (Srdjevic and Srdjevic 2008). Along the northern 30 km of the main canal water is of the high quality and mostly used for regular agricultural irrigation and occasional irrigation of the lowland forest stands along the canal. As to the further downstream, and particularly along the last 15 km before the Danube River, it is not always possible to guarantee even a minimum ecological flow of 0.5 $m^3/s$ during the summer season, and the outflow can be heavily polluted.

Besides the direct users of the water and the operating authorities, inhabitants along the canal, NGOs, ecologists and media also take a keen interest. The system is important for drainage (both natural and wastewater), industrial water supply (sugar industry), irrigation of agricultural and forest areas, fishing and recreation activities. The system includes set of canals and pumping stations.

The MODSIM model (Labadie 2005) network representation of the Nadela system is shown in Figure 7. The model operates on a monthly temporal timescale, and spatially encompasses the entirety of the Nadela river and canal system. For each month, the system's description (configuration and capacities, storage and non-storage characteristics), complete hydrology and the preference structure of imposed demands are modelled as a closed network and solved as a linear programming problem. Local (monthly) optimal solutions and other necessary data are used to define the initial conditions for the next month, and for solving the new linear programming problem.

MODSIM outputs values for the water demands and shortages, as well as frequencies of system failure to meet pre-specified flow and storage targets. These outputs are then available to users to assess possible operating strategies. Temporal scaling in this example involves a multi-year time horizon with a basically deterministic approach in monthly time steps. The water demands are known, which allows the provision of monthly forecasting. Therefore, overall diversions of water can be confidently determined for prescribed priority schemes. On the other hand, the spatial scaling (cf. Figure 7) appears to be too coarse to include additional fine tuning in structuring the system network and developing changes to system operational strategies. For example, an increase or decrease in the number of canal sections, junction and diversion points, or changes in priority schemes and operating rules would be exceptionally difficult to

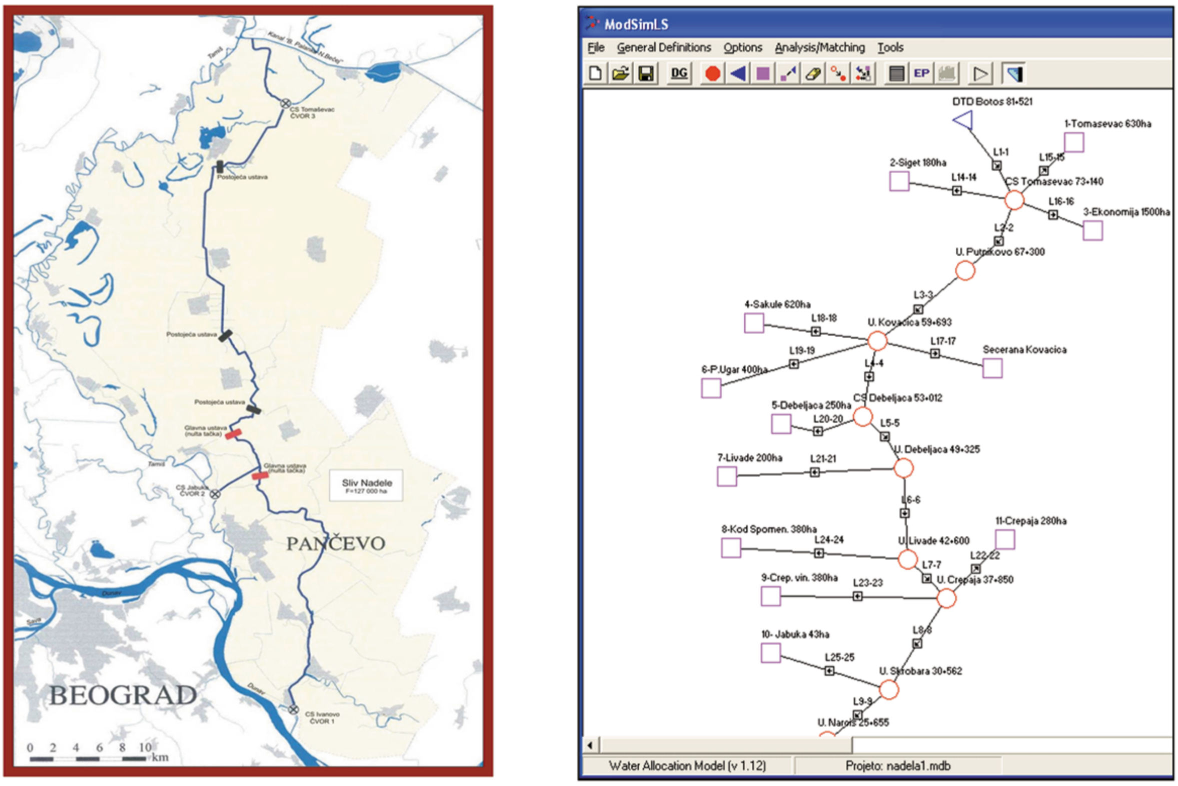


**Figure 7.** MODSIM network for Regional hydro-system Nadela

administer in the MODSIM outputs. This is considered to be a problem common to all complex simulation models. Further examples of how outputs of simulation models can be complicated to administer and manipulate may be found in Srdjevic et al. (2004) where decision-makers were interested in tackling system reliability, resiliency and vulnerability (Hashimoto et al. 1982). These well known conflicting system performance parameters could be incorporated after specific clustering of decision elements.

*No scaling applied*

Yield tables have been traditionally used by foresters to predict forest yields basically in terms of basal area, average or dominant height, wood volume and wood assortments at the hectare/stand scale. Tables have dominated forest management planning and decision making for more than 200 years (Paulsen 1976). In some countries, yield tables are still the only tool to implement the theory of the stand growth into practice (Mihov 2000). Yield tables are a good example representing the absence of scaling related issues as far as spatial scale is a constant and importantly, that time is not a factor (a table can be represented at different temporal resolutions but the underlying model is unchanged). A major problem in the application of yield tables in contemporary forestry practice is that they mainly represent the average development of a ‘normal’ or ’ideal‘ even-aged monospecious stand.

The relevance of yield tables as an instrument to study and simulate forest stands’ growth and development has been subject for many discussions in the recent decades. The main considerations have been related to the need for a better response to changing user needs and environmental conditions, including a substantial shift from the prevalence of pure to mixed stands, changing treatments of stands, which differ significantly from the treatment most yield tables are based on, a change from even-aged to uneven-aged management methods, and changing growth conditions (Nagel et al. 2002)

*Only spatial no temporal*

In contrast to yield tables, the single-tree growth models do not model average parameter values of pure, even-aged stands but study the growth and development of the individual trees. Development of single-tree growth models has been induced by changes in forest management priorities such as the shift from single species, even-aged stands to mixed, uneven-aged ones. In determining the values of the individual tree parameters, tree growth models consider the competitive relationships with the surrounding trees. In this respect, they introduce a spatial dimension (Hasenauer 2005).

The variables inventoried and information required in the most tree growth models include species, height, dbh, age, height to the live crown, crown width, tree coordinates etc. Site factors such as elevation, slope and aspect, depth of soil and humus horizons, relief, soil moisture class and vegetation type are usually included in the growth equations. Although the actual modelling unit in tree growth models is the individual tree, the aggregation of the information obtained provides a more accurate result at an upscaled level (e.g. stand, enterprise, etc.) than information derived by other low resolution models. Such processes of information aggregation (upscaling) are usually obtained by the implementation of growth simulators based on the individual tree models. Implementation of growth simulators ensures an integration of tree models into practice in the future and a closer cooperation between end users (all stakeholders in the forestry chain) and model developers. Through simulators, individual tree growth models may serve as tools for the development of silvicultural scenarios and strategies, transfer of silvicultural strategies into the field, short and mid-term planning on an enterprise level, visualization at a landscape level and many other issues set by forest management practice (Schmitd et al. 2006). The accuracy of information delivered as a growth simulator output depends upon the conventional statistical information input into the implemented model equations. Characterization of validation material and statistical validation procedures may also help users decide which simulator to use. If the forest growth simulator contains random variables, predictions may give different results even with all other conditions being equal. Therefore, information on the randomly controlled model components should be provided (Pretzsch et al. 2006).

The rather “big jump” from the traditional yield tables to individual tree growth models has been met with enthusiasm as well as scepticism. Yield tables are easy to use, and they are “robust”, which means that predictions are usually found to be sufficiently reliable, provided the yield table predictions are calibrated with the measured inventory data. On the other hand, growth estimates for individual trees have many sources of error and simulated graphics may create a false impression that all the predictions are accurate. Comparisons of the actual growth with the predictions are rare, especially at the extremes of density, age and growing site; the few existing ones give rise to scepticism (Windhager 1999). For this reason, Zhang et al. (2009) developed a flexible growth model for beech, which represents a compromise between the inflexible but trusted yield tables and the more sophisticated but error-prone individual tree models. The future lies in compatible models linking high and low resolution models (Cao and Baldwin 1999)

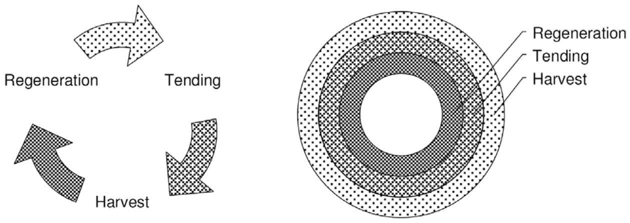
*Only temporal no spatial*

Temporal scaling is a key issue in the risk modelling. Relatively short-term extreme events may change the overall long-term strategy for forest management. To ensure that forest management scenarios are realistic, greater emphasis is usually placed on modelling of potential hazards. The hazard potential includes all the potential threats within a given hazard domain. Potential hazards, their factors and risks can be assessed using probability models as well as in monetary terms (Gadow 2000). For example, Gonzįlez et al. (2006) developed an empirical model for the 12-year probability of fire occurrence in Catalonian stands. Temporal aspects can be seen in reversing the risk assessment modelling to survival modelling; Kouba (1989) used the Weibull distribution for modelling tree survival rate based on stand age. The probability of surviving several hazard factors (survival rate) is multiplicative to surviving each single hazard factor. Such an approach enables the consideration of many risks in forest management planning. However, even the best theory implemented in a computer system does not itself guarantee good management practice, and also very often similar forest ecosystems have large differences in the structural diversity and variability (e.g. Laarmann et al. 2009).

*Both temporal and spatial*

Silvicultural systems integrate appropriate timing, sequence and type of treatments to produce a predetermined set of desired outcomes from a continuous community of trees on a certain area, assuming that the area is sufficiently uniform in composition, structure, age distribution, spatial arrangement, site quality etc. Therefore, silvicultural systems represent both temporal and spatial dimensions of scaling. Historically, they have been developed as a tool to assure predictable crops of commodities throughout a particular period of time, and the rotation period. Appropriately elaborate systems also sustain some important non-market values. Silvicultural systems as management tools are comprised of various activities and operations. The sequences and continuation of operations determines the temporal scaling issue. Activities, which take place during the rotation period of a silvicultural system, are grouped into three main phases or component treatments; regeneration, tending and harvest.

Sequences and continuation of phases and the timing and intensity of actual activities and treatments differ considerably based upon the silvicultural system applied. In even-aged systems, the main component treatments are distinctly separate (clearcutting) or slightly overlapping in time (shelterwood). The regeneration of an age-class/stand is followed by tending during intermediate ages and finally by its harvest again at maturity. Figure 8(A) depicts the interdependence of these phases and the necessity of assuring an integrated approach to management throughout the rotation period. In general, the sequence of treatments in uneven-aged silvicultural systems resembles those in the even-aged ones. Stands managed under uneven-aged systems however differ in terms of their spatial structure (spatial scaling issue). They consist of numerous age classes/cohorts (normally at least three), trees of different ages being dispersed (single-tree selection method) or arranged in groups (group selection method) over the stand area. Periodically, part of the growing space is allocated to a new age class. All the component treatments are simultaneously performed across the stand (Figure 8(B)) during the cutting series. In principle, long-term structural stability is assured in stands managed by the selection system.



**A. Even-aged systems** **B. Uneven-aged systems**

**Figure 8.** Sequences and continuation of phases in two alternative silvicultural systems (clearcutting and selection system), depicting both the independence and continuous nature of these component treatments

The disadvantage of the silvicultural system concept from a management perspective is that it is rather theoretical and generalized. The information is provided and the sequence and intensity of treatments are planned and elaborated at a stand level scale. In practice, site conditions and tree vegetation attributes usually vary widely across stands. Proper management in such cases would require that information is provided and treatments are performed at a bio-group rather than at a stand scale. This consideration would require incorporating more than one silvicultural system concept across the stand.

*Sequence of processes in terms of spatial and temporal resolution*

The sequence in the management planning process includes the identification of landowner objectives, an inventory of resources, the development and implementation of the management strategy to be used and a periodic re-evaluation of the strategy applied. The most important part of this process involves definition of the landowner objectives. This phase determines the planning activities: what resources are to be inventoried, what products and amenities are to be obtained, how to achieve the owner's goals etc. In commercial management planning, primary stakeholders are forest owners, including the state and its institutions, private owners and municipalities. Secondary stakeholders are the local population and community, the wood processing industry, nature and wildlife protection non-governmental organizations, tourists, hunters and anglers etc. Knowledge of the owner's goals is crucial in determining the necessary level and accuracy of the information to be gathered. Forest management plans are based on and are limited by what is biologically/ecologically feasible on an area, what is economically and organisationally acceptable, and what is socially and politically desirable. Modern forest management planning strategies emphasize the 'multiple-use' concept. This promotes not just timber and other raw material production but also the conservation of species and wildlife habitats (Söderbergh and Ledermann 2003) at both the stand and forest enterprise level. It is the level on which management planning is elaborated that determines the scale and accuracy of information needed. After adequate calibration procedures, contemporary growth simulators may provide accurate and sufficient information for the single stand up to whole enterprise level management planning (Pretzsch et al. 2006), provided they are not limited to the upscaling of error-prone individual tree models (Windhager 1999).

## Discussion and conclusions

Different end users (researchers, managers, policy people, non professional, etc.) may look at the different parts of a system. These differences in their interest are often related to scaling issues. Researchers may be interested in a single particular process, or be limited to a particular study site. Their focus is on understanding the system, rather than on a specific output. Indeed, the most interesting parts of a modelling study are often when the output does not match observations, as discovering the reason 'why this is so' leads to new knowledge. On the other hand, managers are concerned with the whole of their enterprise, and seek to have firm, quantified information and projections but may not understand or care about how they were derived. Therefore, it is incumbent on model developers to be clear about the appropriate scale their model can be used on, and on end users to assure that they apply the model correctly. The same applies to policy makers or non-professional model users: all too often the results of limited local studies are used to justify national or international decisions, or generic broad-scale results are inappropriately applied to local decision-making. Model precision and generality are certainly inter-related: high precision is often achieved at the cost of low generality and *vice versa*. The choice of modelling approach is usually a compromise between precision and generality. A good compromise has no straightforward answer and, therefore, the question needs to be addressed to the specific modelling issue and end user needs.

Models are often limited in scale by the availability of information either due to data shortage or limits to our conceptual understanding of ecosystem processes (Kangur et al. 2007). Although we have discussed a few of the methods available for upscaling or downscaling information, there is often a price to be paid in precision. The temptation may be present to accept 'best estimates' as model inputs, but it must be recognised that this could severely affect the precision or accuracy of the outputs. Therefore, model validation is required whenever there are changes in the scale or quality of the input data and assumptions. Defending the use of possibly dubious input data on the basis that "it is the best that is available" is not acceptable, unless validation against field data shows that the imprecision of the inputs does not negate the results of the modelling. Unfortunately, there will be often cases where there is simply not enough data to carry out a desired modelling operation, and modelling based on a 'best guess' of input parameters may in fact be little improvement over simply providing a 'best guess' of the outputs. Model end users are generally not aware of the data rescaling and estimations within a model, and, therefore, may give the results more credence than they deserve. It is the modeller's responsibility to ensure that this does not happen.

The end user needs can be met easily in the decision support systems for forest management. The MOTTI system (Hynynen et al. 2005) is an excellent example used in Finland by forest owners to simulate forest growth and production at the stand and estate level. Models operate mostly on the single-tree level but outputs are designed to give aggregated ecological and economical predictions.

Where models are developed to answer a particular management or policy need, the first question must be "At what scale is the result of this modelling to be applied?", and the second "Is adequate input data available at this scale?" If the end user is unable to answer the first question, then it may be likely that the results will be applied inappropriately. If the answer to the second question is 'no', then data collection is an inescapable first step in the modelling operation.

Forest models are required for a variety of purposes. **Foresters** use them to project the dynamic development of a managed ecosystem and to assess the timber product yields for specified harvesting options in selected stands. **Ecologists** need models to study mutual feedbacks between specific organisms and their environment. **Politicians** require models to evaluate impacts, for example, to predict the effects of a change in climate, or a fundamental change in silvicultural policy, on the development of a strategic resource. The **timber industry** needs them to project availability of raw materials for wider geographical regions.

Models may be used alternatively as diagnostic, simulation or forecasting tools, each use having different input requirements, caveats and risks. Models may also be used as a communication tool, to present common information to a diverse range of stakeholders and enable cooperative policy development. But also next to the model based approach modelling can be used to define users following the purposes for which user groups require information/modelling: (i) Models as research tools, (ii) Modelling for management application and (iii) Modelling as a policy related information tool.

Top-down modelling approaches use broad-scale input values; bottom-up models, however, assume homogeneity at the scale of their input data, and each site is often assumed to be representative of a larger area. At a broad scale outputs should be compatible, where sufficient samples are available for statistically valid estimates in the top-down modelling and for adequate differentiation of heterogeneous sites in the bottom-up approach. For single samples, however, comparisons of results between approaches are likely to be meaningless. The point of model convergence is thus dependent on the scale of the comparison.

In the practice of forestry modelling, it is often necessary to produce meaningful outputs on the basis of rather weak information. Therefore, spatial and temporal scaling issues are an integral part of linking end user needs with our conceptual understanding of ecosystem processes and input and output data issues. Required state variables may be not available and such problems have to be circumvented by a smart conceptual approach or reasonable assumptions. This flexibility calls for an integrated approach and clearly addresses scaling questions since the different levels of detail must be compatible to ensure a consistent modelling output.

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# ОПРЕДЕЛЕНИЕ ПОТРЕБНОСТЕЙ КОНЕЧНЫХ ПОЛЬЗОВАТЕЛЕЙ ЛЕСНОГО МОДЕЛИРОВАНИЯ НА РАЗНЫХ УРОВНЯХ

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*Резюме*

Модели лесных экосистем являются диагностическими инструментами для оценки и понимания экосистемных процессов. Для достижения сбалансированного моделирования необходимо учитывать такие конфликтные интересы как простота, наблюдаемость и биологический реализм. Так как полевые наблюдения часто доступны только в течение коротких периодов времени или в ограниченном пространстве, модели имеют большое значение для экстраполяции в пространстве и времени. Успешное моделирование основано на поиске подходящего масштаба (уровня), но часто ограничено наличием исходных данных. Зачастую часто приходится выдавать результат на основании скудной информации. В этой статье мы обсудим вопросы пространственных и временных уровней в рамках моделирования. Рассмотрим проблемы кросс-масштабирования, их соответствие необходимым данным, а также определение и реализацию конкретных потребностей конечных пользователей.

**Ключевые слова**: лесное моделирование, масштабирование, конечный пользователь