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Approaches for Estimating Critical Loads of Nitrogen and Sulfur Deposition for Forest Ecosystems on U.S. Federal Lands

Linda H. Pardo

Abstract

Projected emissions of sulfur and nitrogen are expected to have continuing negative impacts on forests, in spite of reductions in sulfur emissions as a result of SO_2 control programs. Sulfur and nitrogen emissions present serious long-term threats to forest health and productivity in the United States. This report is intended to explain the differences in approaches for calculating critical loads for forest ecosystems in Europe, Canada, and the United States; it is directed to air quality regulators and Federal Land Managers (FLMs) in the United States, and addresses concerns particular to U.S. Federal lands. The paper describes the basic mass balance approach for calculating critical loads, presents the various critical thresholds, and explains the assumptions inherent in the calculation and data selection procedure. The input necessary from FLMs in the process of estimating the critical load is described.

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Contents

Introduction 1
Purpose 1
Critical Load History 1
Types of Approaches for Critical Load Calculations 4
Steady-state Approach: In-depth Review 7
Base Cation Depletion 7
Nitrogen Saturation 8
Calculation of Critical Loads 8
Similarities and Differences in Approaches 11
Critical Thresholds Related to Ecosystem Effects 11
Thresholds for Estimating Acceptable ANC Leaching 11
Note on Thresholds for Aquatic Critical Loads: ANC 13
Thresholds for Estimating Acceptable N Leaching 13
Thresholds for Estimating Acceptable Soil N Accumulation 14
Summary 14
Data Necessary for Critical Loads Calculations 14
Atmospheric Deposition 16
Forest Type/Nutrient Removal 16
Soil Mineral Weathering 17
Leaching Losses 17
Additional Data Issues 17
Selecting the Method 17
Selecting the Critical Threshold 17
Selecting the Data Method/Models 18
Comparing Results 18
Next Steps: Toward National Scale Critical Loads Calculations 19
Acknowledgments 20
Literature Cited 20
Additional Web Resources 24
Glossary 24

INTRODUCTION

Purpose

The objective of this paper is to provide resource managers an overview of the critical loads approach for estimating acceptable levels of sulfur (S) and nitrogen (N) deposition to forest ecosystems. Acceptable levels of deposition will protect vulnerable ecosystems and allow restoration of impaired ecosystems (Blett 2004). Projected emissions of sulfur and nitrogen are expected to have continuing negative impacts on forests, in spite of reductions in sulfur emissions as a result of SO_2 control programs. Sulfur and nitrogen emissions present serious long-term threats to forest health and productivity in the United States. This paper is intended to explain the differences in critical loads approaches used in Europe, Canada, and the United States for forest ecosystems; it is directed to air quality regulators and U.S. Federal land managers (FLMs) and addresses concerns particular to U.S. Federal lands.

The focus of this paper is on the steady-state mass balance methods that have been used most broadly to calculate critical loads. Many researchers have calculated critical loads using variations of the same approach. This paper seeks to clarify the differences in those approaches. The two key variations are in selecting the critical threshold (a particular pH, for example) and in dealing with missing data. These topics are addressed in Sections 3 and 4. The paper describes the basic mass balance approach for calculating critical loads, presents the various critical thresholds, and explains the assumptions inherent in the calculation and data selection procedure. The input necessary from FLMs in the process of estimating the critical load is described. Critical loads for aquatic ecosystems are not addressed in this paper.

Critical Load History

Background

The critical load approach has been used in Europe to identify levels of deposition expected to cause harmful ecological effects. The United Nations Economic Commission for Europe (UNECE) convention on Long-range Transboundary Air Pollution (LR TAP) (U.N. Econ. Com. for Europe 2007) resulted in maps of critical loads being generated for Europe by 1995 (Posch et al. 1995). Those maps have been modified (Posch et al. 2001) and the resulting critical loads have been used to regulate emissions through the 1999 Gothenburg Protocol via the UNECE Convention on LRTAP. Projected emissions reductions in Europe of 63 percent for SO_2 and 41 percent for NO_x by 2010 are estimated to protect 78 million hectares, reducing the area of exceedance from 93 million hectares in 1990 to 15 million hectares by 2010 (U.N. Econ. Com. for Europe 2000).

Critical Load Definition

The critical load is the level of deposition below which significant harmful ecological effects do not occur (see Sidebar 1). This definition is used in Europe (UBA 1996). A current definition used by FLMs is “the threshold deposition of pollutants at which harmful effects on sensitive receptors begins to occur” (Blett 2004). Both of these definitions indicate that to protect ecosystems deposition must be below the critical load; they are, therefore, essentially interchangeable. Another definition was used in the FLMs Air Quality Related Values Work Group (FLAG) Phase One program (National Park Service 2000) that “the critical load is the level of deposition above which detrimental effects occur”. In this case, the critical load is the maximum acceptable deposition level. For example, using the

Sidebar 1.—Roles of Federal Land Managers and Research Scientists in Setting Critical Loads

Defined by researchers with direction from FLMs

Detrimental, Harmful—used to describe a condition of the ecosystem where normal nutrient cycling processes have been disrupted in such a way that the ecosystem can no longer maintain its health and productivity. For pristine class I areas, this may also include change from "natural" (pre-industrial) condition.

Defined by FLMs with input from researchers

Significant—this is a decision by the FLM about what sort of effect is considered large enough to be of importance. For example, what would be considered a small change (not significant) in an intensively managed ecosystem, may be considered significant in a class I area.

Acceptable—Acceptable levels of deposition are levels that will protect vulnerable ecosystems and allow restoration of impaired ecosystems.

Sidebar 2.—Time and the Critical Load

The concept that the critical load is a steady-state property can be confusing. This means that the critical load is based on the capacity of the ecosystem to buffer deposition inputs. Inherent in the critical load calculation is the assumption that all the processes in the ecosystem are at steady state, and that the critical load is, therefore, not a function of time. An example of steady state is a bathtub with the water running and the drain plug open, with the flow set so that the level of the water is unchanging (also see Fig. 2.A).

The steady-state assumption means that fewer data are necessary to calculate the critical load. Alternatively, if we have enough information to describe the rates of the physical, chemical and biological processes, we can estimate the time until a given condition is reached. This process is dynamic modeling, which can yield valuable information about time to damage and time to recovery if the necessary input data are available. Time to recovery (and to damage) are important factors in policy and management decisions and are used to set target loads rather than critical loads.

first two definitions, if the critical load is 10 kg/ha/yr, that means that if the deposition is 10 kg/ha/yr, it will harm the ecosystem. Using the third definition, if the deposition is 10 kg/ha/yr, the ecosystem will be protected. For reasons of scientific clarity and legal defensibility, it is important to be consistent about how the critical load is defined. Because of the historic use of the first definition in Europe, and because of the way empirical critical loads are set (based on the deposition level where detrimental effects are observed), it is preferable to define the critical load using either of the first two definitions—that the critical load is above the acceptable level of deposition.

Target Load Definition

The critical load is a value that is based on scientific information about expected ecosystem response to a given deposition level. The target load is set by policy makers (land managers or air regulators) to meet their objectives. In selecting the target load, the policy maker may consider economic cost of emissions reductions, timeframe, and other matters. The target load may be higher than the critical load. For example, the target load may be chosen to protect 95 percent of forest lands, as has been done in Europe. The target load may be lower than the critical load if a very sensitive area is to be protected in the short term, generally, in cases where current deposition exceeds the critical load. For federal resources, the target load would be somewhat below the critical load to be consistent with federal resource protection mandates (Porter et al. 2005). For areas where the critical load has been exceeded, air regulatory agencies may choose an "interim target load" higher than the critical load, as a progress goal toward the critical load (Porter et al. 2005, Skeffington 1999). The process for selecting the critical load and target load is illustrated in Figure 1. The critical load is not a time-dependent value; the target load, in contrast, can take into account the timeframe for a desired ecosystem condition or recovery (see Sidebar 2).

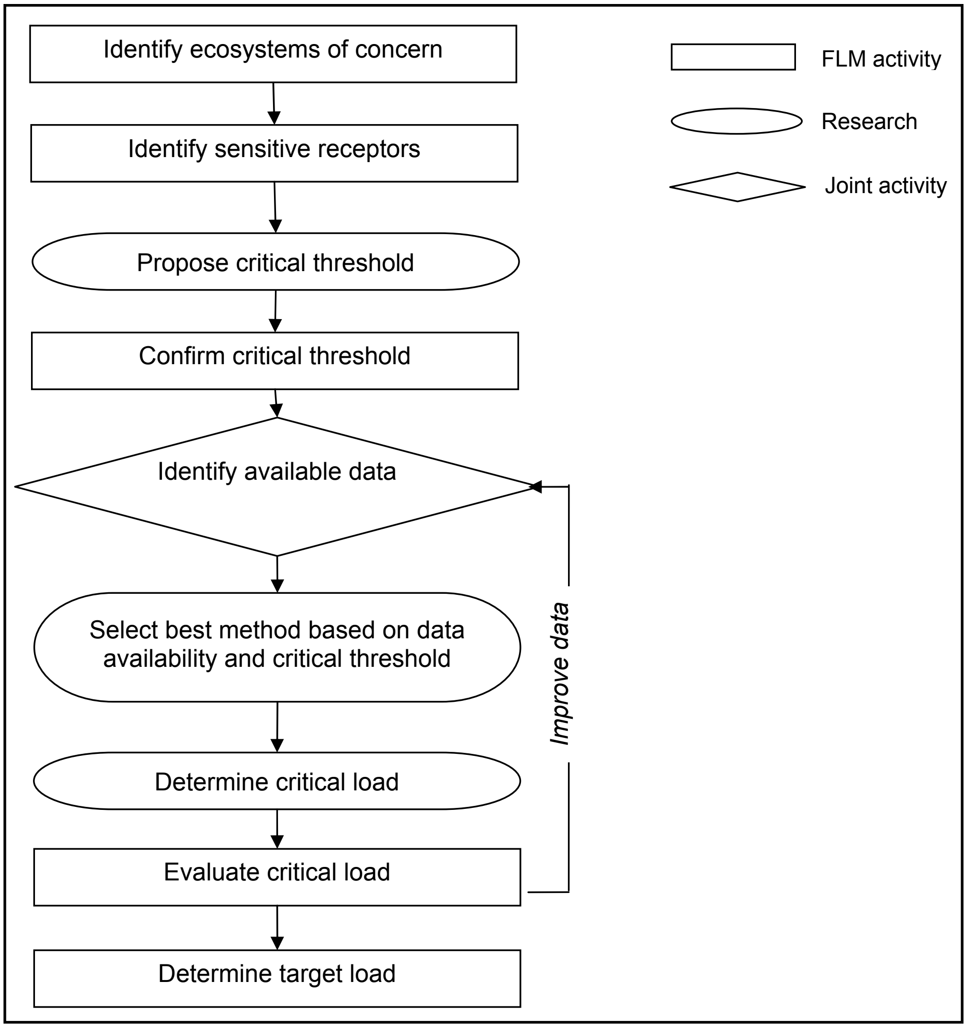

Figure 1.—Flowchart of FLM involvement in the process of calculating critical loads.

Procedure and roles

The application of critical loads requires interaction between policy makers, air quality regulators, federal land managers, and scientists. The policy makers create the context for enforcement and select the target loads or interim target loads based on the critical loads that the research scientists have calculated. Federal land managers (and air quality regulators) interact with the scientists calculating critical loads at several points (Fig. 1). First, they identify the regions of concern. Next, they identify any particularly sensitive receptors (or areas of an ecosystem) of concern; they may be called upon to provide data for critical load calculations. Finally, given the critical load calculations, they determine the target load.

The research scientists calculating critical loads identify the appropriate critical threshold to protect the sensitive receptor or area of an ecosystem. They assemble the data available for the sites, then they use the best available method (based on the data availability) for calculating the critical load. In some cases, a range of critical loads may be calculated to reflect the heterogeneity of the site or uncertainty in calculation methods. In these cases, scientists work with the FLM to determine the most appropriate critical load to address the FLMs' goals. It is the manager's role to determine what change is unacceptable and the role of research scientists to determine at what level of deposition that change is likely to occur.

Inputs from FLMs

To be sure that critical loads are calculated and presented in a way that is most useful, FLMs' input is necessary at several stages in the process (Fig. 1). Calculating critical loads requires that assumptions be made to convert available site data into the simple form required for the critical load calculations. These assumptions may include utilizing data from other sites or from the literature when site-specific data are not available. It may also include calculating averages of vegetation or soil data, since many sites include multiple species and some include multiple soil types.

Federal land managers' input is necessary prior to making critical load calculations. Federal land managers need to determine the spatial extent of the site for which the critical load is to be calculated. A site could be a plot, a stand, or a watershed. The scale of the site has a significant impact on the assumptions necessary to calculate the critical load. Federal land managers also must decide whether the critical load should be presented as a mean for the area or should be given as a range of values from the least to most sensitive areas. And, if the critical load is presented for the most sensitive area, what is the scale on which that is defined? A watershed, a stand, a plot, a tree? Further, the FLM may identify certain sensitive species of concern, as the critical load would normally be calculated for the existing forest type (based on a mean of the species present). This broad approach could overlook a species with greater sensitivity. The same is true for sites with mixed soil types: generally the critical load would be calculated based on the area-weighted average of the two soil types. If one soil were considerably more sensitive, the calculated critical load using the mean soil type might be too high and not protect the sensitive soil type. Finally, the FLM can assist in the selection of sites that are representative of the ecosystems of concern for which critical load calculations would be made.

Types of Approaches for Calculations of Critical Loads

There are three main approaches for calculating critical loads. **Empirical approaches** are based on observations of response of an ecosystem or ecosystem component (e.g., foliage, lichens, soil) to a given, observed deposition level. Empirical critical loads then are calculated for the site where the data were obtained and, generally, they are applied to similar sites where such data are not available.

Simple mass balance approaches are based on estimating the net loss or accumulation of nutrients based on inputs and outputs of the nutrient of concern (e.g., base cations, nitrogen). Simple mass balance methods are steady-state models that calculate the critical load of deposition to an ecosystem over the long term (i.e., one rotation in land managed for timber, 100+ years in wilderness). They are used at sites with moderate to intensive data availability.

Dynamic models use a mass balance approach expanded by incorporating internal feedbacks—such as accumulation of N in the system, or exchange of base cations between soil and soil solution from year to year. Dynamic models can predict time to damage and time to recovery. Dynamic models are typically used at sites where intensive data are available.

Data availability drives the selection of the type of approach for calculating critical loads. For sites where little or no data are available, empirical approaches must be based on data from similar or comparable sites. For sites with a moderate to intensive level of data available, simple mass balance approaches are used. Most dynamic models may be used only at sites with substantial data that, generally, must range over some period of time. However, a highly tested dynamic model can be subsequently applied to an adjacent region with more sparse data (e.g., Chen and Driscoll 2005). Each type of critical load calculation may yield a different critical load value for the same site because of the different assumptions involved. Each type of critical load may be calculated for the overall ecosystem or for a particular ecosystem component (if data for that component are available).

Field observation-based approaches

Empirical critical loads are determined by using literature or field observations of detrimental ecological effect and noting the deposition level at which the effect occurred. The lowest deposition level at which a response occurs is considered the critical load. In cases where there is a variation in the level of deposition that causes a given response, a range for the critical load often is reported. An example of empirical critical loads that are widely applied is the European summary of empirical critical loads for N (Bobbink et al. 2003). This paper includes a table of ecosystem types and their respective empirical critical load for N. The utility of empirical critical loads is that they can be used to determine the critical load based on the best available information for that ecosystem type when data are not available at a given site.

Advantages of empirical critical loads are simplicity and ease of use (they require no calculation), and applicability over a broad set of conditions. Empirical critical loads also can be set for different ecosystem types if data are available. The main conceptual advantage of empirical critical loads is that, under the best circumstances, they link ecosystem response to deposition. They are particularly useful for setting critical loads for nutrient N, which are difficult to model.

Disadvantages of empirical critical load calculations are primarily due to a lack of quantitative understanding of the empirical observations. This approach is based on observed cause and effect responses rather than understanding of a process or mechanism. Thus, it is difficult to be certain of the level of deposition that causes a given response. Further, since the observations of ecological response to a given deposition level are based on past scenarios, they may not show the breadth of response possible in the future. For example, if a response is observed at a given deposition level (or fertilization level) after a certain number of years, it is possible that a lower deposition over a longer period of time would cause the same detrimental effect. This would mean that the empirical critical load was too high. For example, N deposition of 25 kg/ha/yr for 7 years may cause excess nitrate leaching, and therefore, the critical load would be set at 25 kg/ha/yr. However, at another site (or even the same site), N deposition of 15 kg/ha/yr for 15 years might cause the same response.

Another pitfall of the empirical approach is that the observed response may be unique to the site at which it was measured (because of particular site history, soil thickness, etc.) and may not be representative of the ecosystem type in general. In this case, the critical load might be too high, which would put sensitive areas at risk. On the other hand, the critical load might be too low, which would not be of great concern for class I areas where the mandate is to err on the side of protecting the land. Class I areas include national parks, wilderness areas, monuments, and other areas of special national and cultural significance above a size criterion and in existence as of August 7, 1977. They are protected, under the Clean Air Act, more stringently than other areas.

In Europe, empirical critical loads have been proposed both for acidity (UBA 2004, Chapter 5) and for nutrient N (Bobbink et al. 2003, UBA 2004). In the United States, because of the nonlinear behavior of acid-base processes, and heterogeneity of ecosystems, it is less useful to propose empirical critical loads for acidity. For nutrient N, however, it is useful to estimate empirical critical loads for different categories of ecosystems. Several recent summaries of effects of N deposition on ecosystems provide useful information about possible empirical critical loads for nutrient N (Aber et al. 2003, Bobbink et al. 2003, Driscoll et al. 2003a, Fenn et al. 2003a, 2003b). These assessments are being compiled to generate empirical critical loads for N for ecoregions in the United States (Pardo et al., in press).[1] For some ecosystem types where data for calculating critical loads are sparse (e.g., arid ecosystems), it may be advantageous to estimate empirical critical loads based on the best understanding of impacts in those or similar ecosystems elsewhere, rather than to attempt to model them.

Steady-state Approaches

Steady-state approaches are based on scientific understanding of ecosystem processes. These approaches also are called simple mass-balance methods. They utilize the best available data for estimating the net loss or accumulation in the ecosystem of the nutrient of concern.

The advantage of mass balance equations is that they are scientifically based on the mass balance concept—that if you deplete the ecosystem pool of an essential

[1]For further information see http://nrs.fs.fed.us/clean_air_water/clean_water/critical_loads/.

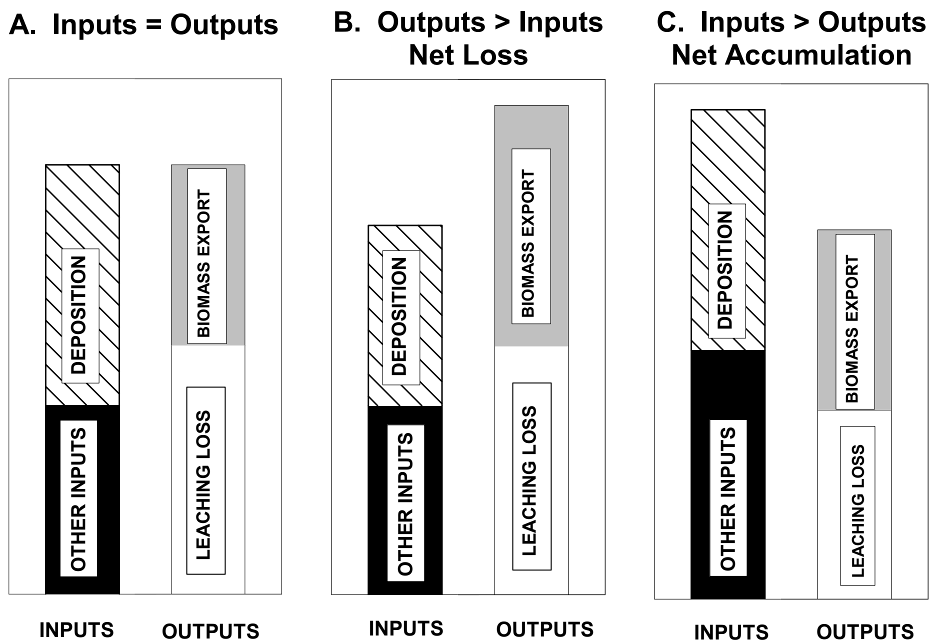

Figure 2.—Mass Balance Approach. This is a conceptual schematic for any quantity of interest. Specific examples are shown in Fig. 3. Other inputs refers to any inputs in addition to deposition, such as mineral weathering or nitrogen fixation. Biomass export refers to removal by harvesting (which is zero for class I areas) and loss from fire. Leaching loss refers to export in solution from the soil; for a watershed this would include stream export.

nutrient, such as base cations, ultimately it will harm the forest (Fig. 2). By contrast, if you have too large a net accumulation of N in the system, it will be detrimental for the forest ecosystem. The mass balance is based on ecosystem processes and can be applied at different sites, taking into account site condition.

The disadvantages of the mass-balance approach are the requirements for extensive data that are not readily available; considerable uncertainty about the critical thresholds for ecosystem response used; and lack of information about time until the expected ecosystem response or the timeframe for recovery. Further, steady-state models cannot describe any individual site well, as they do not incorporate sufficient detail or include ecosystem processes.

Nonetheless, simple mass-balance models often are considered the best available model for estimation of critical loads across large regions. These approaches may provide the maximum return for the level of information they require. This is the approach that was used to generate maps for Europe (Posch et al. 2001).

Dynamic Modeling Approaches

Dynamic modeling approaches were developed to address the inability of steady-state models to assess time to ecological damage or time to recovery. These dynamic models give a more realistic representation of change in an ecosystem by taking into account that response to one change does not occur immediately throughout the whole ecosystem. Instead, a change in deposition can cause a change in chemistry over some period of time; this change in chemistry causes a subsequent change in biology over time.

The advantages of dynamic models are that they include a more realistic representation of the complexity of ecosystems and that the results allow assessment of the time for a particular ecosystem effect to occur. Many of the impacts associated with air pollution involve these nonsteady-state conditions; recovery from these impacts also involves nonsteady-state conditions.

The disadvantages of dynamic models are that the models may have very large data requirements, take a fairly long time (and high level of expertise) to apply to a given site, and that some models may describe a particular ecosystem type or site very well, but may not be applicable to a wide variety of sites. A general difficulty in dynamic modeling is finding the balance between having the model describe a particular system very well (which dynamic models may do, but steady-state models cannot) and having the model be useable at a broad range of sites. Typically, dynamic models have not been applied over broad areas for terrestrial ecosystems. One exception is the Very Simple Dynamic model being developed in Europe (http://www.pbl.nl/en/themasites/cce/methods_and_models/vsd-model/index.html), which is discussed on page 19. Dynamic models may be applied over broad areas with little data, but not without compromising the quality of the results. In such cases, as in any calculation of critical loads, better quality and finer resolution of input data will yield more accurate results.

Since the critical load concept is essentially a steady-state concept, an appropriate application would be to combine a steady-state estimation of the critical load with a dynamic modeling estimation of time to damage or recovery (see Sidebar 2). In this case, it is important to ensure that the assumptions and conditions of the dynamic model are consistent with those of the steady-state model.

STEADY-STATE APPROACH: IN-DEPTH REVIEW

This section explores the steady-state approach in more detail, because it is the methodology most likely to be used initially by FLMs based on the level of site data generally available. The steady-state approach also is called the simple or steady-state mass balance approach. It is based on the concept that if outputs from an ecosystem exceed inputs, you will have a net loss from the ecosystem (Fig. 2). On the other hand, if inputs to the ecosystem exceed outputs, you will have a net accumulation in the ecosystem (Fig. 2C). Simple mass balance methods are steady-state models that calculate the critical load of deposition to an ecosystem over the long term (i.e., one rotation or several cutting cycles, depending on the management, or 100+ years for wilderness).

The main inputs to the ecosystem include atmospheric deposition and weathering of mineral soil; the outputs from the ecosystem include leaching losses and biomass removal (Fig. 2A). Leaching losses may be estimated from water moving downward through the soil or, for watersheds, from export of the stream draining the watershed. To calculate a critical load, one needs to define the ecosystem of concern. In this paper, we explain calculating the critical load for the forest ecosystem—from the trees down through the soil. Although we discuss various conditions in one part of the ecosystem or another, we do not calculate a critical load to protect that component, but to protect the whole forest ecosystem including that component. (Protect, in this context, means to prevent harmful ecological effects.)

Base Cation Depletion

One result of acidic deposition is the acidification of soils and surface water (Driscoll et al. 2001, Reuss and Johnson 1985). As acidic deposition moves through the ecosystem, there is exchange between the acidifying hydrogen ion (that entered the ecosystem with sulfate and nitrate) and base cations (calcium (Ca), magnesium (Mg), sodium, (Na), potassium (K)). When sulfate and nitrate exit the ecosystem, they take these base cations with them, leaving the hydrogen ion (H+) in the ecosystem. There are several consequences of the removal of base cations from the ecosystem and the accumulation of H+ in the soil, soil solution, and surface water: the acidification of soils and surface water; the potential mobilization of aluminum (Al), which can be toxic for plants and animals; and the net loss or depletion of base cations, which are essential plant nutrients.

These three chemical changes then impact the biological components of the ecosystem. There can be direct effects of the increased acidity (Driscoll et al. 2001). The most broadly described concern is that from Al toxicity for aquatic (Cronan and Schofield 1990, Driscoll et al. 1984, Driscoll et al. 2003b) and forest (Cronan and Grigal 1995, Sverdrup and Warfvinge 1993b) ecosystems. The depletion of base cations is of increasing concern (Driscoll et al. 2001, Federer et al. 1989). Without an adequate supply of base cations, trees are susceptible to toxicity from Al in excessively acidic environments and from nutritional deficiency of base cations. Consequences of base cation depletion include reduced cold tolerance and increased winter injury (DeHayes et al. 1999, Schaberg et al. 2000), crown dieback (Wilmot et al. 1995), increased susceptibility to pest and disease (McLaughlin and Wimmer 1999), reduced regeneration, reduced growth and increased mortality (Schaberg et al. 2002), and species composition changes.

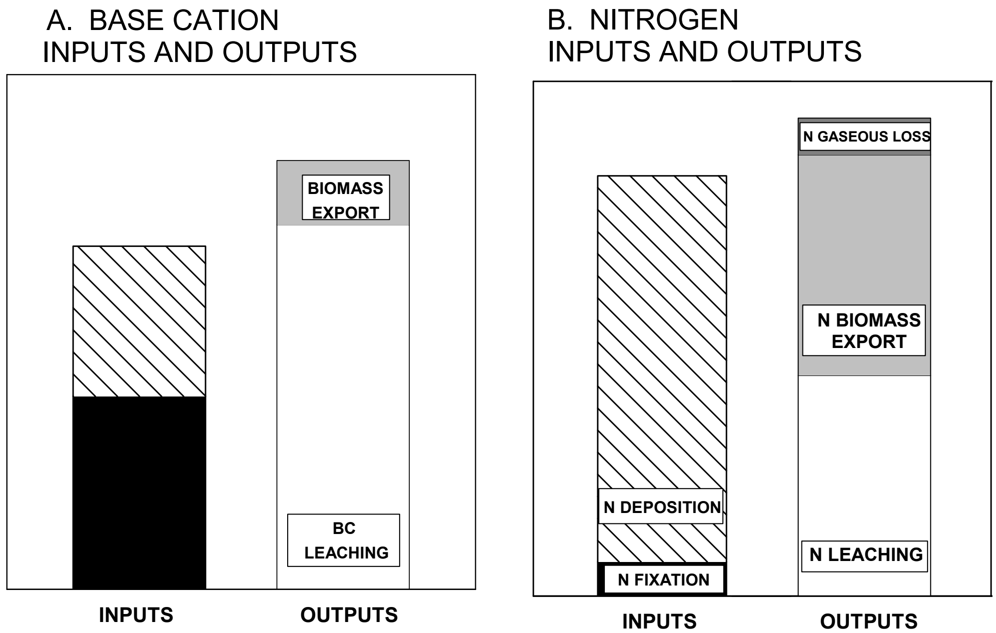

Figure 3.—Mass Balance Approach: A. Base Cations (BC), and B. Nitrogen. Two examples of how the mass balance approach uses various types of site-specific data on chemical and biological inputs and outputs. In both cases, outputs are greater than inputs; the difference is used in the calculation to determine whether critical loads have been exceeded. A. In this example, there is a net loss of BC (outputs>inputs). B. In this example, there is also a net output of N from the ecosystem. Biomass export could be via fire or harvesting.

To calculate a critical load for acid deposition inputs (or acidity), we calculate a mass balance on the base cations necessary to neutralize the acid input. If we revise Figure 2 in the context of base cation mass balance, the inputs are base cation (BC) deposition and soil mineral weathering; the losses are BC in biomass removed and leaching (Fig. 3A).

Nitrogen Saturation

In addition to nitrogen's role as an acidifying compound, there is concern about excess input of nitrogen in the ecosystem and N saturation. This excess nitrogen initially will accumulate in soil and subsequently be lost via leaching. While increased nitrogen may increase productivity in many terrestrial ecosystems (which are typically N limited) this is not necessarily desirable in protected ecosystems, where natural ecosystem function is desired. Excess nitrogen can lead to nutrient imbalances in trees, understory species, nonvascular plants (lichens), or mycorrhizal fungi; changes in composition of these species; and ultimately to declines in forest health (Aber et al. 1998, Driscoll et al. 2003a, Fenn et al. 2003b).

If we revise Figure 2 in terms of N saturation concerns (Fig. 3B), the inputs for the mass balance are atmospheric deposition of N, N fixation (where significant), and the N losses are via leaching, biomass removed, accumulation of N in soil, and gaseous losses of N (denitrification and volatilization, where significant).

Calculation of Critical Loads

Federal land managers need values for critical loads for N and S deposition. To determine the critical load for N and the critical load for S, we need to determine the critical load for N and S inputs combined (i.e., for the total acid inputs to the ecosystem). Once the critical load for S+N is determined, we must apportion that critical load into a S portion and a N portion—in other words, to determine how much of the acid input may enter the ecosystem as S and how much as N. The N input component is the sum of ammonium and nitrate deposition. Once the critical load for S+N is calculated, there are many possible combinations of N and S deposition that would be acceptable (Sidebar 3).

Sidebar 3.—Critical Loads and Target Loads for Acidity, Sulfur, Nitrogen, N Nutrient

The ***critical load*** is the level of deposition below which no harmful ecological effects occur.

The ***target load*** is the level of deposition set by policymakers to protect a given area of sensitive ecosystem components. The target load may be higher or lower than the critical load based on considerations of economic cost of emissions reductions, timeframe, and other matters.

Critical loads and target loads can be calculated for different inputs: acidity (N and S combined), S, N, or N nutrient (to address detrimental effects caused by excess N and N saturation).

The ***critical load for acidity*** is the total deposition of acidity inputs to the ecosystem (S and N combined); it is sometimes referred to as critical load S+N.

The ***critical load for S*** is the total deposition of S to the ecosystem; when no critical load for N is calculated, the critical load for S is equal to the critical load for acidity.

The ***critical load for N nutrient*** is the deposition of N to an ecosystem below which no harmful effects of N saturation occur.

The ***critical load for N*** is the total deposition of N to the ecosystem. The critical load N is either simply the critical load for N effects from acidification which is calculated by subtracting the S deposition from the critical load acidity. Preferably, a critical load N nutrient is calculated and the critical load N used will be the lower of the two.

One source of confusion in calculating critical loads comes from the fact that N deposition has two main consequences: it can acidify the ecosystem, and it can lead to plant nutrient imbalances. Sulfur deposition, in contrast, only acidifies the ecosystem. Determining the critical load for N, therefore, is a three-step process. First, we calculate the critical load for N to address concerns of acidification, then we calculate the critical load for N to address concerns of excess nutrient N. Finally, we use the lower value as the critical load for N.

Process for Calculating the Critical Load for S+N (Acidity)

Calculating critical loads for S and N requires several steps. First, we calculate the critical load for acidity—for the total S + N deposition (note that this is the value people use when they calculate a critical load only for S—actually, they calculate the critical load for acidity).

The next step is to calculate a critical load for N nutrient. This is the level of N deposition that would lead to plant nutrient imbalances and, in some cases, to changes in species composition and biodiversity. In some cases, the critial load for N nutrient is much lower than the critial load for acidity (Fig. 4A). In those cases, the critial load for N nutrient is taken as the critical load for N, and the critical load for S is calculated by taking the critial load for N+S and subtracting critial load for N (so the total S and N critical loads add up to the critial load for for S+N). When the critial load for N nutrient is high (Fig. 4B), and when the critial load for N nutrient is greater than the critial load for acidity (Fig. 4C), then the calculation of critial load for S and critial load for N is more complicated. In these cases, there are many possible combinations of S and N deposition that would be permissible (combine to equal the critial load for S+N; Fig. 4B and C).

Process for Estimating the Critical Load for N Nutrient

The critical load for N nutrient is sometimes referred to as the critial load for N eutrophication. However, the term critial load for N nutrient is usually preferred because eutrophication implies a concern with effects only in aquatic ecosystems. In many calculations for critical loads, the critical load for N nutrient is

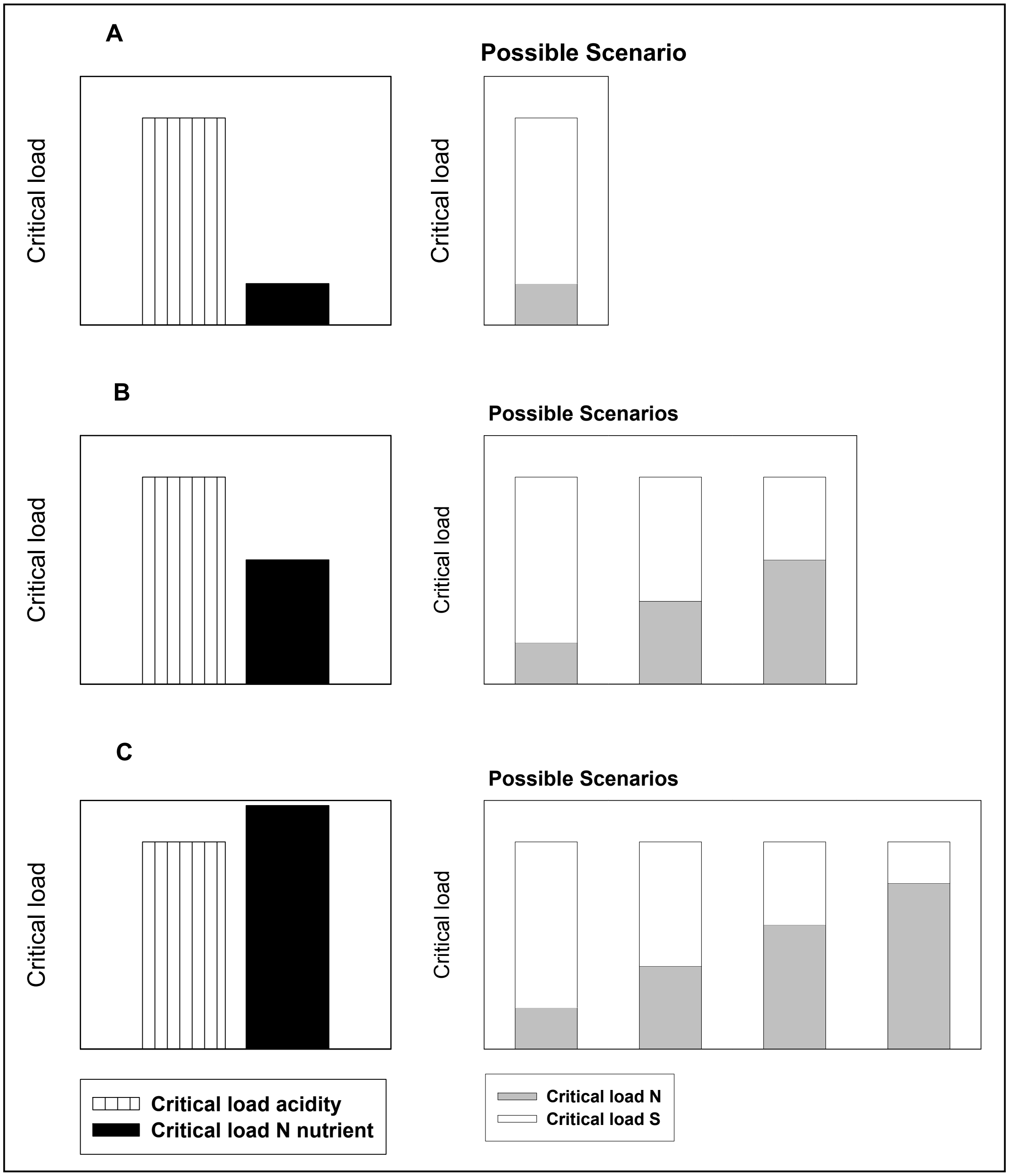

Figure 4.—Calculation of critical loads for S and N is a muti-step process. Critical loads are calculated separately for acidity and nutrient N in forest ecosystems (boxes on left). The calculated critical load for acidity can be made up various proportions of N:S deposition (boxes on right). The critical load for N should not exceed the critical load for nutrient N. (A) critical load for N nutrient is much less than critical load for acidity; in this case, the critical load for N nutrient is significantly less than the critical load for acidity, so the critical load for N is set to equal the critical load of N nutrient. Once the critical load for N is defined, then the critical load for S is simply the difference between the critical load for acidity and the critical load for N. (B) critical load for N nutrient is similar to critical load for acidity; in this case, there are many possible combinations of acceptable values for the critical load for N and the critical load for S, as long as the critical load for N does not exceed the critical load for N nutrient. Some of the possible combinations are shown. (C) critical load for N nutrient is greater than critical load for acidity; in this case, there also are many possible combinations of acceptable values for the critical load for N and the critical load for S; any combination of N and S that adds up to critical load for S+N is acceptable. The upper limit for critical load for N is the critical load for S+N. Some of the possible combinations are shown.

overlooked. Initially, this occurred because of a lack of understanding about the potential harm that excess N deposition could do to ecosystems, and because concerns were only about acidification and focused on sulfur. With SO_2 abatement legislation, sulfur emissions and deposition have decreased significantly (Butler et al. 2001, Driscoll et al. 2001). This decrease in S, combined with increases of N deposition in some places, has made N increasingly significant (Fenn et al. 2003a). Another reason that critical loads for N nutrient were not initially calculated is that internal cycling of nitrogen makes the effects of N deposition difficult to predict and model. However, it is very important that the critical load for N nutrient be calculated. In some locations, consequences of N deposition are far more significant than from acidification (Fenn et al. 2003b).

Similarities and Differences in Approaches

It is critical for FLMs to understand when using and evaluating critical loads that all of the steady-state approaches are based on a mass balance on the forest ecosystem (Fig. 2). Fundamentally, the approaches are all the same: they compare inputs to and outputs from the ecosystem. However, the approaches differ in two ways: in determining the critical thresholds and in estimating and modeling data that are not available. The following two sections address these issues in detail.

CRITICAL THRESHOLDS RELATED TO ECOSYSTEM EFFECTS

One of the challenges in estimating critical loads is relating the ultimate biological or ecosystem effect to some measurable quantity—often a chemical characteristic. This chemical characteristic is referred to as a critical threshold. For aquatic ecosystems, this can be more straightforward—a given pH or acid neutralizing capacity (ANC) can cause a detrimental response in a certain population. In terrestrial ecosystems, setting critical thresholds is more difficult because of the complexity of nutrient cycling and the spatial heterogeneity of these ecosystems (UBA 2004). Different critical thresholds will lead to different critical loads. In general, the critical threshold that best addresses the receptor of concern is selected (for example, for some receptors a decrease in pH would be the problem, for some receptors, an increase in the Al concentration). A receptor might be a particular organism (salamander, lichen, tree species) or it might be an ecosystem compartment (soil, trees, etc.). When several critical loads are calculated using different receptors, the lowest value is generally used as the critical load. This is a key point for FLMs: their input is needed on what ecosystem effects are of particular concern and whether there are specific receptors of exceptional concern within the ecosystem.

The following sections detail the different parameters that have been used to set critical threshold values. The main values that need to be set in the mass balance equations are: (1) acceptable ANC leaching rate; (2) acceptable N leaching rate; and (3) acceptable N accumulation rate in soil. A more detailed, technical explanation can be found in the Mapping Manual (UBA 2004, Section 5.3.2.2).

Thresholds for Estimating Acceptable ANC Leaching

One of the critical thresholds in the mass-balance equation used for critical load calculations is the acceptable ANC leaching rate—or the loss of acid neutralizing capacity that is acceptable over the long term. In aquatic systems, the acceptable ANC concentration may be set directly because data are available that link ANC concentration directly to indicator organism health. In terrestrial ecosystems, however, organism or ecosystem health are not linked directly with ANC concentration, but are related to other chemical parameters such as pH or Al that are linked to ANC leaching. These chemical parameters (e.g., pH, Al concentration) can be used to calculate ANC leaching, so if the chemical parameter is set to a given value (the critical threshold), the value of ANC leaching calculated based on the critical threshold is the acceptable ANC leaching rate.

To determine what that ANC value would be, we must take into account information about soil pH and exchange between Al and H+. The procedure for determining the acceptable ANC leaching rate is to identify the sensitive receptor, select a chemical criterion,

Table 1.—Critical thresholds used to calculate acceptable ANC leaching rate

Critical threshold	Typical value	Receptor	Ecological response	Comments
Aluminum concentration	0.2 meq/L	soils		Protection of drinking water (groundwater) to EPA standard of 5-20 μeq/L
BC:Al ratio	1 mol/mol	soils	fine root damage	a more conservative value is 10
Ca:Al ratio	1 mol/mol	soils	fine root damage	a more conservative value is 10
pH	4.0	soils	loss of Al from soil complex; BC depletion	4.2 also is used
Base saturation	No decrease	soils	further BC depletion; subsequent nutrient deficiency	Implies that current condition is acceptable

and finally, determine what the critical threshold for that criterion would be based on past research and understanding (Aherne and Farrell 2002). Typical values for critical thresholds are given in Table 1.

Aluminum Concentration

The main critical thresholds that have been used are aluminum concentration or the base-cation-to-aluminum ratio, BC:Al, in soil solution. At certain levels, Al can be toxic to plants (Sverdrup and Warfvinge 1993b). There is tremendous variability in how ecosystems actually respond to given Al concentrations, making it difficult to develop an Al threshold. Ecosystem response depends heavily on site condition, which can vary widely: one ecosystem well below the threshold may appear to thrive while another may show signs of clear decline (Cronan and Grigal 1995). All of the equations involving a critical value for Al rely on a defined relationship between Al and H^+ concentration based on equilibrium with the mineral gibbsite ($Al(OH)_3$). A fixed ratio of Al to H^+ is used to predict Al concentration based on pH. This value is determined observationally or from the literature (Table 1). Because these data are not widely available, a single literature-derived value is usually used (UBA 2004, Table 5-11, p.V-22).

Base Cation to Aluminum Ratio

Aluminum concentration has been used extensively as a critical threshold in aquatic ecosystems. But in terrestrial ecosystems, it has been observed that the abundance of Al relative to base cations (BC) is more important. For example, Al values may be high, but if BC values are also high, plant health is not compromised. In contrast, relatively low Al values may be detrimental if BC values are very low. In this case, the acceptable ANC leaching rate is determined by the critical BC:Al ratio and the gibbsite ratio of Al:H^+ (Table 1).

Calcium to Aluminum Ratio

Research (summarized by Cronan and Grigal 1995) has suggested that the most significant ratio is that of calcium to aluminum, and not simply the ratio of total base cations to aluminum. In this case, a critical threshold just for Ca:Al ratio is used. Thus, the acceptable ANC leaching value is determined by the critical Ca:Al ratio and the gibbsite ratio of Al:H^+.

One complicating factor in determining what that critical Ca:Al ratio should be is the range of ecosystem response to a given Ca:Al ratio. Recent research has made some advances in explaining the variability in tree response to high Ca:Al in soil solution. Instead of measuring total foliar Ca to determine whether a tree has a Ca deficiency, as was done in the past, researchers have separated out functionally important Ca, known as membrane-associated calcium or mCa (DeHayes et al. 1999). The variability in tree response to low Ca:Al in soil solution is greatly reduced when the functionally significant plant Ca pool is measured rather than the largely structurally bound total Ca. Further, this research suggests that the Ca content of the plant prior to exposure may control the plant's susceptibility to Al toxicity—plants with high levels of Ca may sequester the toxic Al, minimizing any detrimental effects on the plant (Borer et al. 2004). While this mCa value is useful for explaining the relationship between soil solution Ca:Al ratio and plant Ca deficiency (and subsequent declines in plant health), the data are difficult to obtain, and, therefore, not widely available.

pH

For soils high in organic matter, using a criterion based on H^+ concentration is recommended (UBA 2004). In this case, the acceptable ANC leaching value is determined by the critical pH (or H^+ concentration) and the gibbsite ratio of Al:H^+. Because organic matter forms compounds with Al in solution, the gibbsite ratio must be adjusted for these types of soil (Hall et al. 2001, van der Salm and de Vries 2001; see UBA 2004, Table 5-11, p. V-22). If the gibbsite ratio is not modified, the calculations will overestimate available toxic Al and give a falsely low value for the critical load.

Base Saturation

To move away from the threshold concept, the New England Governors/Eastern Canadian Premiers (NEG/ECP) Forest Mapping Group decided the critical criterion would be no change in base saturation (NEG/ECP 2001). This criterion avoids a precise threshold as a basis for setting the critical load. Base saturation (BS) is a measure of the available (exchangeable) BC in the soil system as a fraction of the maximum BC potentially available. High base saturation (greater than about 20 percent) means that base cations are abundant; low base saturation (less than about 10 percent) means that base cations are less available and the soil is more susceptible to detrimental effects from acidic deposition.

This approach may appeal to FLMs, because it is based on limiting human-induced change, which may be more aligned with the FLMs' mandate than selecting a particular critical threshold. The base saturation approach, however, is not without complexity. Using "no decrease in base saturation" as the criterion does not necessarily give you equivalent results in terms of ecosystem condition. For example, if you have two ecosystems, one with a BS of 25 percent and one with a BS of 4 percent and you prevent a decrease in BS, you end up with two ecosystems that meet the criterion, but clearly these ecosystems are not equally buffered. The ecosystem with the BS of 25 percent is much less susceptible to detrimental effects from acidic deposition than the ecosystem with the BS of 4 percent. However, from the FLM perspective, with the focus on preventing this change to the ecosystem, the base saturation criterion may be adequate. On the other hand, this approach does not account for any loss of base saturation due to acidification in the past, it simply prevents further loss.

Note on Thresholds for Aquatic Critical Loads: ANC

The following section is included for information and to compare aquatic with terrestrial critical loads. For aquatic ecosystems, the critical threshold is the ANC, which is set directly (rather than calculated, as above). Generally, the acceptable ANC value ranges from 0-50 µeq/L (see Pembroke 2003, DuPont et al. 2005, UBA 2004, section 5.4.1.4, pages V-32-V-33). Lower values of ANC (0 or 20 µeq/L) usually are set to protect against chronic acidification; higher values of ANC (50 µeq/L or in some cases higher) are usually set to protect against episodic acidification.

Thresholds for Estimating Acceptable N Leaching

The acceptable N leaching rate is the allowable loss of nitrogen from the ecosystem, usually via streamwater. Like the acceptable ANC leaching rate, this rate is determined to be acceptable over the long term. Excess leaching of N is a sign of N saturation (Aber et al. 1989, Stoddard 1994). This is a value used in critical load equations and is not a measured value. Typical values for critical thresholds are given in Table 2.

Nitrate Concentration

The critical threshold for nitrate concentration exported from the ecosystem may range from 0.2-0.4 mg N/L. At those levels, nutrient imbalances have been observed in coniferous and deciduous forest stands (van Damm 1990). To determine the critical leaching rate, the critical concentration is multiplied by the actual stream flow rate.

Nitrate leaching

The nitrate leaching term also may be determined directly, if it is not possible to determine it using the critical nitrate concentration. Typically values ranges from 0-5 kg N/ha/yr, depending on stand type, age, and management (Gundersen 1992). For example, acceptable

Table 2.—Critical thresholds for nitrogen

Critical threshold	Typical value	Receptor	Ecological response	Comments
Nitrate concentration in soil solution	0.2-0.4 mg N/L	coniferous, deciduous forests	nutrient imbalances	
Acceptable N leaching (soil solution)	0-5 kg N/ha/yr			higher N leaching can indicate N saturation
Acceptable soil N accumulation	0.2-1 kg N/ha/yr	soil	N leaching, acidification	a more conservative value is 0

N leaching rates would increase in the following sequence: tundra < boreal coniferous < temperate coniferous < temperate deciduous (Grennfelt and Thörnelöf 1992).

Thresholds for Estimating Acceptable Soil N Accumulation

The acceptable N accumulation is the amount that can be added to the soil N pool each year without ultimately leading to elevated N losses. This value varies widely as a function of soil type and age, species composition, and past land-use history; it is difficult to determine. Values typically used for the threshold for acceptable soil N accumulation range from 0.2-1 kg N/ha/yr, although some conservative estimates set the value at 0 (Table 2).

Summary

The difficulty in identifying an abiotic threshold value that will lead to a given biological concern is caused by lack of understanding of the mechanisms that regulate ecosystem dynamics (processes), and also, in cases where the mechanism is understood, the extreme variability in response of different ecosystems to the same deposition level. Also, for acid-sensitive sites it might be difficult to separate out N nutrient effects from N acidification effects. This points out an important scientific gap in understanding the mechanisms of detrimental effects of atmospheric deposition on forest ecosystems. If deposition is extremely high (such as 150 kg N/ha/yr), we know we will see a response, and if deposition is extremely low (1 kg N/ha/yr) we know we will not see a response. The task at hand, then, is to figure out when we will start seeing effects, if deposition ranges approximately from 5-25 kg N/ha/yr. This is not to say that there are not biological measures of damage from atmospheric deposition, but that the measures in forest ecosystems tend to be of fairly severe damage. In the United States, ecosystems usually experience moderate levels of deposition. An early indicator that would tell us that the trees are at risk, not that they are already severely damaged, would be more useful to land managers.

DATA NECESSARY FOR CALCULATIONS OF CRITICAL LOADS

The data used to calculate critical loads can have an impact on the critical load value calculated. Therefore, in some cases, it may not be legitimate to compare results that are based on different data assumptions. The most significant variation between different applications of the steady-state mass balance method is in the selection of the critical threshold. The second most significant factor is how the inputs and outputs of the mass balance are estimated. Which data are used is generally a function of data availability: when measured data are available, differences in critical loads between sites should primarily reflect actual differences in characteristics of each site and not incorporate the uncertainty of modeled input data. Measured data, however, may include variation resulting from differences in measurement method; often such differences may be systematic. When measured data are not available, the inputs need to be estimated either by category or modeling. In these cases there are many different approaches that can be used. The data requirements include information about soil, vegetation, deposition, and surface waters/leaching. The data requirements are summarized in Table 3.

Table 3.—Summary of data needed for calculations of critical loads using the steady-state mass balance approach. Data from individual sites are usually preferable to modeled or extrapolated data.

Site description data	Mandatory	Optional	Atmospheric/climate data	Mandatory	Optional
Latitude and longitude	x		Deposition (N, S, Ca, Mg, K, Na)	x[a]	
Elevation	x		Precipitation volume (long-term)	x[a]	
Size of site		x[b]	Mean annual temperature (long-term)	x[a]	
Land-use history		x[b]	Mean annual evapotranspiration	x[a]	
Disturbance history		x[b]	Runoff	x[a]	
County		x[b]			
Soil data			Tree and forest data		
Number of soil pits/site	x		Stand composition	x	
Mineralogy	x[a]		DBH (diameter at breast height)	x	
Soil profile descriptions		x[c]	Nutrient concentration (N, Ca, Mg, K)		
Soil depth		x[c]	by biomass fraction by species		x[c]
Soil texture		x[c]	Annual biomass removal rate		x[c]
Soil bulk density		x[c]	Biomass by species		x[c]
Course fragments		x[c]	Average height by species		x[b]
Parent material		x[c]	MAI (mean annual increment)		x[b]
Organic matter percent		x[c]	Volume		x[b]
Soil series		x[b]	Nutrient uptake by species		x[b]
Extractable nutrients (Ca, Mg, K, Na)	x[d]	x[b]	Nutrient ratios		x[b]
Cation exchange capacity	x[d]	x[b]			
Base saturation	x[d]	x[b]			
Volumetric soil moisture		x[b]			
Lysimeter data, DON, DOC		x[b]			
pH		x[b]			
Forest health data			Stream water data		
Foliage transparency		x[b]	Size of catchment		x[b]
Crown density		x[b]	Stream water chemistry		x[b]
Dieback		x[b]	Stream water flux		x[b]
Insect damage and disease		x[b]			
Areas of forest decline		x[b]			
Vegetation structure		x[b]			
Other (lichens, …)		x[b]			

[a]Data may be modeled in some regions if models are available
[b]Data helpful for interpreting results
[c]Data may be calculated or come from (regional) values reported in the literature
[d]Necessary for making critical loads calculations using the criterion of "no decrease in base saturation"

It is important to note that in Europe there are some significant differences (from the U.S. scenario) that drove the development of calculation methods for critical loads. First, in terms of vegetation, the stands typically are much more homogeneous—fewer species, many more single-species stands, many plantations. A second difference is that most stands in Europe are intensively managed and harvested. Wilderness on the scale of the United States does not exist in Europe. This means the data assumptions may be more straightforward in Europe and data may be more readily available (species composition, stand age, rotation length, etc.). In all cases, the better the quality and reliability of the data used in the critical loads calculations, the more reliable the results. This holds true for dynamic models as well as steady-state models.

The purpose of this section is to explain how the different assumptions used in assembling data can cause differences in critical loads values and to present some of the most common approaches for estimating data. The subsequent sections address the major components of the critical loads calculations.

Atmospheric Deposition

Atmospheric deposition data are often available for wet deposition, but infrequently available for dry deposition, and rarely available for cloud/fog inputs. Also, dry deposition is highly uncertain, spatially heterogeneous, and rarely tested using multiple approaches. (Even when measured values are available for dry deposition, models need to be used to transform the measured concentrations into loads of dry deposition, and these models may not be parallel). Our purpose here, however, is not to present an exhaustive list of the differences in measurement and estimation methods, but to give an overview of the types of approaches used to determine the atmospheric deposition that is necessary for calculating the exceedance of the critical load. The exceedance of the critical load is the actual deposition minus the critical load.

Two approaches are used to estimate total deposition in the absence of measured values: fixed ratios of total deposition:wet deposition and modeling. Using fixed ratios is generally discouraged by atmospheric deposition specialists, however, it is very simple and therefore, practical. Ratios could be based on CASTNet (Clean Air Status and Trends Network; http://www.epa.gov/castnet) and NADP (National Atmospheric Deposition Program; http://nadp.sws.uiuc.edu) data. Note that fixed dry:wet deposition ratios may not be realistic, at least for the northeastern United States, because decreases in SO_2 concentration appear to be reducing the dry:wet deposition ratio (Chen and Driscoll 2004). The modeling approach can be very useful (e.g., ClimCalc) (Ollinger et al. 1993), in that one can generate a value at any site regardless of whether any data exist for that site. Other regional models include a model of wet deposition for the eastern United States (Grimm and Lynch 2004), and of deposition in the Rocky Mountains (Nanus et al. 2003). Such models are usually regional and vary in their ability to capture extremes in deposition (as a function of elevation, distance from edge, aspect, and inclusion of cloud and fog inputs). These models may have numerous drawbacks, however, including time period modeled or data too sparse to interpolate well between measurement points.

Approaches that do not adequately account for dry, cloud and fog inputs will result in an underestimation of the exceedance: not all sites where the actual deposition exceeds the critical load will be identified. Some method of estimating total (wet + dry + cloud) deposition is therefore needed to interpret critical loads meaningfully. Fixed ratios will underestimate deposition when cloud and fog inputs are very high, probably at sites where inputs occur in a few large events (for example, arid or seasonally dry areas). Fixed ratios may overestimate deposition in some deciduous stands, high precipitation areas, and low-lying areas. Fixed ratios are also problematic when there is a shift in vegetation from hardwood to conifer or if there are marked variations in elevation. If fixed ratios are based on S deposition, they will not be adequate for predicting N deposition.

In summary, it is important to estimate deposition as accurately as possible, because underestimation will cause too few susceptible sites to be identified (sites where the actual deposition exceeds the critical load).

Forest Type/Nutrient Removal

In forests that are harvested, it is necessary to define a term for biomass removal. In class I or wilderness areas, the biomass removal term would include only biomass removed by fire. Note that large-scale forest disturbances, such as blow-down, insect, or disease outbreaks, would not be treated as biomass removal because the bulk of the biomass remains on the site. It is also necessary to determine the forest type for estimating total deposition and for identifying susceptibility.

The nutrient removal term can include considerable uncertainty in the species composition, estimated biomass, estimated nutrient concentration, and estimated biomass removed. Estimating the nutrients removed with biomass requires first determining the biomass removed and then the concentration of nutrients in the biomass (which is a function of species composition).

Biomass Removal Estimates
If forest harvesting rates are unknown, it may be possible to obtain county level information and use that as a crude estimate of biomass removed. Actual biomass records that can be extrapolated to the future are preferable.

For class I or wilderness areas, it is necessary to have information about the frequency and intensity of fire.

Nutrient Concentration Estimates
To estimate nutrient removal, the nutrient concentration is multiplied by biomass removal (generally by tree compartment: stem, branch, bark, foliage). For the northeastern United States, Pardo et al. (2005) developed a tree chemistry database to use when site-specific data are not available. It might be a useful step to develop such a database for the whole country.

If biomass removal is overestimated, the critical load for acidity will be too low, and the critical load for N nutrient will be too high (not adequately protective).

Soil Mineral Weathering

Mineral weathering is very significant for sites where there are concerns about acidification because of the role weathering plays in buffering acidic inputs. However, there is considerable uncertainty in estimating weathering rates. There are several different approaches for estimating the mineral weathering. The most sophisticated is the Profile model (Sverdrup and Warfvinge 1993a), which has extensive data requirements, including a comprehensive measurement of minerals present in each soil horizon (mineralogy). A simpler approach is to assign the soils to categories based on substrate type and to calculate weathering using a regression equation based on soil texture, depth, and mean annual temperature. Finally, in the absence of even soil texture data, the simplest method involves selecting categories of soil weathering based on bedrock type, adjusted for precipitation, vegetation type, soil texture, soil drainage, BC deposition (UBA 2004, Section 5.2.2.).

Underestimating the weathering will cause the critical load to be too low; overestimating weathering will cause the critical load to be high.

Leaching Losses

Leaching loss terms in the critical loads equations (as discussed previously on pages 11-13) are values that are set at a level determined to be acceptable over the long term. These values are not measured data.

Additional Data Issues

Nitrogen fixation inputs and denitrification (gaseous N loss) losses often are assumed to be negligible and therefore excluded from critical loads calculations. However, they should be included at sites where they are significant.

For some problematic land types, these assumptions/calculations break down. For example, if a large fraction of the land is wetlands, ignoring reduction processes and the production of organic acids may be problematic.

Also, lands that were previously in agriculture or experienced a severe fire will retain large quantities of N. For these sites, the acceptable soil N accumulation may be considerably larger than the range reported (Table 2).

SELECTING THE METHOD

Selecting the Critical Threshold

The two factors that drive the selection of the method to use for calculating critical loads are the critical threshold and data availability. The critical threshold is selected based on which receptor is to be protected. In addition, management goals and defensibility may be considered in selecting the critical threshold. For class I or wilderness areas, it is important to identify any sensitive receptors of concern to ensure that the whole ecosystem will be protected. The critical load for each ecosystem component may be different, and there may be ecosystem components more (or less) sensitive than the one for which data are available. If data are not available that link a given receptor to a particular chemical criterion and critical threshold, the critical threshold must be selected from those for which there is information. The critical load would then be presented to the FLM with the information about which receptor was used (soil, etc.). Then the FLM would decide about how to set the target load, based on the sensitive receptor of concern.

Selecting the Data Method/Models

The next step is to determine how the data will be synthesized or modeled to generate the input and output values necessary for the calculations. The goal is to use the best data available at a given site. Therefore, when intensive data are available for a site, those would be used (Table 3). If fewer data are available, the missing data need to be modeled or estimated. In either case, it is necessary to document which default variables were used as input, which site-specific data were used, which data were estimated or extrapolated from other sites, and which critical thresholds were used. Finally, when very few data are available, a categorical approach would be used (see UBA 2004, Section 5.2.2; Pardo and Duarte 2006).

Comparing Results

Several scenarios will result in having multiple critical loads values to compare. Different values for the critical load may be generated if critical loads are calculated for different ecosystem components, different critical thresholds, or for individual parameters (for each tree species, rather than using a weighted mean species composition, or for each soil type, rather than the average soil), using different models to calculate input and output values for the mass balance equation, or using steady-state and dynamic modeling.

Critical Thresholds and Ecosystem Components

One example of different critical thresholds and ecosystem components being used is in comparing the European approach used by the International Cooperation Programme (ICP) on Mapping and Modelling Critical Loads and Levels and Air Pollution Effects, Risks and Trends to the approach used in calculating critical loads for Eastern Canada and New England by the New England Governors/Eastern Canadian Premiers Forest Mapping Project (NEG/ECP 2001). The ICP approach uses the critical threshold of BC:Al in soil solution equal to 1 (mol/mol) to protect roots from Al toxicity, while the NEG/ECP approach uses a critical threshold of no change in base saturation to protect the soil from depletion of base cations. In both cases, the goal is to protect the forest ecosystem. In the ICP case, the receptor is plant roots, in the NEG/ECP case, the receptor is the soil. The NEG/ECP result is more conservative, but may result in greater variation in the ultimate ecosystem condition because of the variation in current base saturation (as discussed on page 13).

Most sensitive receptors

In some cases there is an especially sensitive species. Rather than calculate a single critical load (based on the site average) for a site with multiple tree species, it may be appropriate to calculate the critical load based on that individual species. The critical load for the average, based on the species composition, might not protect that sensitive species. This is another point when input from FLMs is needed. While it can be helpful to have information about the most sensitive tree species, it may not be useful for the FLM to be presented with several critical loads for a site (and if different types of soils are present, the number of critical loads calculated would increase). Therefore, depending on the objective of the FLMs, they may request the critical load for the average conditions, for the most sensitive combination, or for the range.

Federal land managers also need to provide input on the issue of the scale on which the critical load is calculated. Clearly, as discussed previously, the scale on which the critical load is calculated will have an impact on its value. In general, as the resolution gets coarser, the critical load will increase because the small fraction of the most sensitive ecosystems will have relatively less impact than when those ecosystems are evaluated separately.

Dynamic Models

Dynamic models can provide useful information about the time to damage and to recovery. They must be used in combination with a steady-state assessment of the critical load. Available dynamic models are used to evaluate the critical load for acidity.

Dynamic models vary greatly in their complexity and ease of use. Some of these models are simple extensions of the steady-state method, other models are process-based models that incorporate site data, and others are mixed. One of the challenges of using a dynamic model to calculate critical loads for forest ecosystems is to model both the soil chemistry and the forest vegetation growth

Sidebar 4.—Differentiating Critical Loads from Target Loads

The critical load is a property of the ecosystem and is independent of time. A target load can take time into consideration (for example, the time to achieve a certain condition). The critical load does not depend on the current condition of the ecosystem; the target load must take that into account. When "critical loads" are presented for different time periods for an ecosystem, these are actually target loads.

and response accurately. Often a soil model and a tree model are linked. Examples of linked models include PnET-BGC (Gbondo-Tugbawa et al 2001, Gbondo-Tugbawa and Driscoll 2002, Gbondo-Tugbawa et al. 2002), Century-MAGIC, SAFE , and DayCent/PHREEQC.

The Very Simple Dynamic Model (VSD) was developed by the International Cooperation Programme (ICP) Mapping and Modelling Group. The VSD model was intentionally kept simple so that it could be applied at many sites, and it might be an interesting tool for FLMs to use for broad-scale critical loads calculations.

One of the biggest challenges in the forest growth component is accurately capturing the complexity of N cycling. At this point, few dynamic models are used to evaluate the critical load for N nutrient, however PnET-BGC has been used in this way (Backx 2004).

Some caution is necessary in comparing dynamic models with steady-state mass balance models without checking that the assumptions are the same. It also is important to note that the critical load does not vary with time (Sidebar 4). Occasionally, modeling results are presented in which a target load (for achieving a certain condition in a certain period of time) is presented loosely as a critical load.

NEXT STEPS: TOWARD NATIONAL-SCALE CALCULATIONS OF CRITICAL LOADS

While the focus of national critical loads calculations would be to identify the areas likely to be the most susceptible to impacts from atmospheric deposition, it also would be useful for identifying critical gaps in the data and in developing priorities for monitoring at these sites.

Making critical loads calculations on the national scale in the United States would present many challenges. It is critical to develop a protocol for making these calculations so that the process is standardized across the country. Because of the heterogeneity of ecosystems and variation of deposition and because of the variability in data available, this process needs to allow for multiple approaches based on the best available data at each site. In the absence of measured site data, which is usually preferable if the data are of good quality, a protocol would specify the order of preference of different approaches for calculating or estimating values based on ecosystem type and data available.

The European ICP Mapping Manual (UBA 2004) can serve as a model for a standard method for calculating critical loads for the United States (see web resources page). However, alterations to that protocol are necessary to make it suitable for use as a federal land management tool in the United States. These alterations should focus on achieving greater standardization in the process, especially concerning the selection of the critical threshold and the procedure for filling in gaps in data. In Europe, the selection of the critical threshold and procedure for manipulating data to generate the inputs necessary for critical load calculations were at the discretion of each country, and were, therefore, made in many different ways. Such nonuniformity would not be desirable for federal land management in the United States.

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ADDITIONAL WEB RESOURCES

Clean Air Status and Trends Network (CASTNet):	http://www.epa.gov/castnet
Critical load resources for Federal land managers:	http://nrs.fs.fed.us/clean_air_water/clean_water/critical_loads/
Empirical nitrogen critical loads for natural and semi-natural ecosystems: 2002 update:	http://www.icpmapping.org/cms/zeigeBerich/13/related_documents.html
ICP Forests information:	http://www.icp-forests.org
ICP Forests manual:	http://www.icp-forests.org/Manual.htm
ICP Forests reports:	http://www.icp-forests.org/Reports.htm
ICP Forests assessment:	http://www.icp-forests.org/pdf/review.pdf
ICP Mapping and Modelling:	http://www.icpmapping.org
Long-range Transboundary Air Pollution Convention:	http://www.unece.org/env/lrtap
National Atmospheric Deposition Program:	http://nadp.sws.uiuc.edu
PROFILE soil chemistry model:	http://www2.chemeng.lth.se/models/profile/index.shtml
Very Simple Dynamic (VSD) soil acidification model:	http://www.pbl.nl/en/themasites/cce/methods_and_models/vsd-model/index.html

GLOSSARY

Acid neutralizing capacity, ANC—A measure of the ability of a solution to neutralize acid imputs.

Base cations—Calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}) and potassium (K^{+}); elements or ions with a positive charge (cations) that can neutralize acids.

Base saturation, BS—A way of measuring the base cations available to plants. Given as the percentage of potential cation exchange sites that have exchangeable base cations on them and is expressed as a percentage of the total cation exchange capacity.

Categorical approach—Used when very few or no data are available at a site. It is an empirical approach which classifies the site into one of several categories associated with a value or range of values for each parameter in the critical load equation (for example BC weathering, nutrient removal, etc.).

Cation exchange—The interchange between a cation (positively charged ion) in solution and another cation on the surface of soil particles.

Cation exchange capacity, CEC—A measure of the total exchangeable cations that a soil can absorb.

Critical load, CL—The level of deposition below which significant harmful ecological effects do not occur (see Sidebar 1, page 2).

Critical load for acidity—The total deposition of acidity inputs to the ecosystem (S and N combined); it is sometime referred to as the critical load for S+N.

Critical load for N—The total deposition of N to the ecosystem. The critical load for N may simply be the critical load for N effects from acidification which is calculated by subtracting the S deposition from the critical load for acidity. Preferably, a critical load for N nutrient is calculated and the critical load for N used will be the lower of the two.

Critical load for N nutrient—The deposition of N to an ecosystem below which no harmful effects caused by excess N (N saturation) occur. These effects include elevated N availability and leaching, plant nutrient imbalances, changes in species composition, and other detrimental impacts on ecosystem health.

Critical load for S—The total deposition of S to the ecosystem; when no critical load for N is calculated, the critical load for S is equal to the critical load for acidity.

Critical threshold—A chemical characteristic (usually easily measurable) that is related to the ultimate biological or ecosystem effect of concern. For example, in aquatic systems, a given acid neutralizing capacity (critical threshold) may be related to fish mortality (the biological effect of interest).

DBH—Diameter at breast height. DBH is used to measure tree size and to estimate tree biomass.

Denitrification—A microbial process that converts nitrogen to a gaseous form which can then be exported from the ecosystem.

DOC—Dissolved organic carbon

DON—Dissolved organic nitrogen

Dynamic model—Computer models that incorporate internal feedbacks, such as accumulation of N in the system, or exchange of base cations between soil and soil solution from year to year, and allow for the prediction of time to damage and time to recovery.

Empirical critical load—Based on observations of response of ecosystem or ecosystem component (e.g., foliage, lichens, soil) to a given, observed deposition level. Empirical critical loads can be calculated for the site where the data were obtained; usually, they are applied to similar sites where such data are not available.

Exceedance—The exceedance of the critical load is the actual deposition minus the critical load.

FLM—Federal Land Manager

Gibbsite equilibrium—The relationship or exchange between aluminum and hydrogen ion (H+) concentration is based on equilibrium with the mineral gibbsite. A fixed ratio of aluminum to hydrogen ion is used—this ratio is K_{Gibb}. Mathematically, this is expressed as: $[Al^{3+}]_{crit} = K_{Gibb} \times [H^{+}]_{crit}$ The ratio, K_{Gibb}, is used to predict Al concentration based on pH. K_{Gibb} is determined observationally or from the literature.

Mean annual increment—A measure of the net increase in biomass of a tree or forest.

Mass balance—An approach used to determine the status of an ecosystem by comparing the inputs to the system and the outputs from the system. A mass balance can be calculated for any quantity of interest, for example, water, nitrogen, etc.

Sensitive receptor—A part of the ecosystem of concern to FLMs. The sensitive receptor might be a particular organism (salamander, lichen, tree species) or an ecosystem compartment (soil, trees, etc.).

Simple mass balance model—Based on estimating the net loss or accumulation of nutrients based on inputs and outputs of the nutrient of concern (e.g., base cation, nitrogen).

Steady state—A condition of an ecosystem where inputs are matched by output; there is no net change in a system at steady state.

Target load—The level of deposition set by policymakers to protect a given area of sensitive ecosystem components. The target load may be higher or lower than the critical load based on considerations of economic cost of emissions reductions, timeframe, and other matters.

Pardo, L.H. 2010. **Approaches for estimating critical loads of N and S deposition for forest ecosystems on U.S. federal lands**. Gen. Tech. Rep. NRS-71. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 25 p.

Projected emissions of sulfur and nitrogen are expected to have continuing negative impacts on forests, in spite of reductions in sulfur emissions as a result of SO_2 control programs. Sulfur and nitrogen emissions present serious long-term threats to forest health and productivity in the United States. This report is intended to explain the differences in approaches for calculating critical loads for forest ecosystems in Europe, Canada, and the United States; it is directed to air quality regulators and Federal Land Managers (FLMs) in the United States, and addresses concerns particular to U.S. Federal lands. The paper describes the basic mass balance approach for calculating critical loads, presents the various critical thresholds, and explains the assumptions inherent in the calculation and data selection procedure. The input necessary from FLMs in the process of estimating the critical load is described.

KEY WORDS: base cation depletion, nitrogen saturation, simple mass balance model

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