1 INTRODUCTION

L.H. Pardo, C.T. Driscoll, C.L. Goodale

1.1 Objectives

This publication provides a scientific synthesis of the current state of research and knowledge about the response of terrestrial and aquatic ecosystems to nitrogen (N) inputs (N deposition or N additions), and, where possible, identifies critical loads for atmospheric N deposition. It also targets policy makers and resource managers who are seeking a scientific basis for making decisions about the potential broad ecological impacts of air pollution, as well as the impact of specific pollution sources on particular ecosystems.

1.2 Background

1.2.1 Historical N Deposition

Human activity in the past century has led to an exponential increase in N emissions (Galloway et al. 2003). Primary sources of anthropogenic N emissions are high temperature combustion of fossil fuels (i.e., electric power plants, motor vehicles), and the production and use of fertilizers and animal manure (Galloway et al. 2003). Deposition of N has paralleled emissions (Butler et al. 2003; US EPA 2009a, 2009b) so that, in most areas, current rates of N deposition are an order of magnitude higher than pre-industrial levels (Dentener et al. 2006, Holland et al. 1999). Pre-industrial atmospheric N deposition is thought to have been 0.4 to 0.7 kg ha^{-1} y^{-1} (Holland et al. 1999). Contemporary N deposition levels in the United States are on average one-fifth of those observed in Europe (Holland et al. 2005). Wet N deposition reported by the National Atmospheric Deposition Program (NADP) ranges from <1 to >7 kg ha^{-1} y^{-1} for 2008 (NADP 2008), and dry deposition can supply large quantities of additional N, but is far more difficult to quantify (e.g., Dentener et al. 2006). Regional maximum values can be substantially higher than the wet deposition reported by NADP. In the Northeast, maximum N deposition of 30 kg ha^{-1} y^{-1} has been reported for high elevation sites (Miller 2000); in the Southeast, deposition was modeled as high as 31 kg N ha^{-1} y^{-1} (Weathers et al. 2006); and in the Pacific Southwest, where dry deposition and fog events can represent the bulk of N deposition, mean annual N deposition of 71 kg ha^{-1} y^{-1} was reported (Fenn et al. 2008). Nitrogen deposition has increased significantly in the last decade in some regions of the United States and is projected to increase further in certain regions of the country (Lehmann et al. 2005, Nilles and Conley 2001). Estimates of N deposition are discussed in more detail in Chapter 3.

1.2.2 Effects of N deposition

The increases in atmospheric N deposition after the middle of the 20th century initially raised few concerns about detrimental ecosystem impacts. Many terrestrial ecosystems are N limited. Hence, additional N inputs could have a fertilizing effect, which was perceived as beneficial for some ecosystems. Increased tree growth due to N deposition has been demonstrated in northeastern U.S. forests (Thomas et al. 2010). However, substantial debate continues over the existence and magnitude of deposition-induced growth enhancement in forests globally (DeVries et al. 2008, Magnani et al. 2007, Nadelhoffer et al. 1999, Sutton et al. 2008). Furthermore, elevated N inputs can lead to detrimental effects on ecosystems (Nihlgård 1985), including soil and surface water acidification, plant nutrient imbalances, declines in plant health, changes in species composition, increases in invasive species, increased susceptibility to secondary stresses such as freezing, drought, and insect outbreaks, as well as eutrophication of fresh and coastal waters (Galloway et al. 2003). High concentrations of ammonia (NH_3) can be directly toxic to plants (Krupa 2003).

Nitrogen saturation can be defined as the condition when available N exceeds plant and microbial demand. Aber et al. (1989, 1998) described four stages of N saturation in forest ecosystems (Fig. 1.1): background conditions (stage 0), an initial fertilization response (stage 1), flattened response of N mineralization but increased net nitrification (stage 2), and detrimental effects on plant health and growth and general decrease

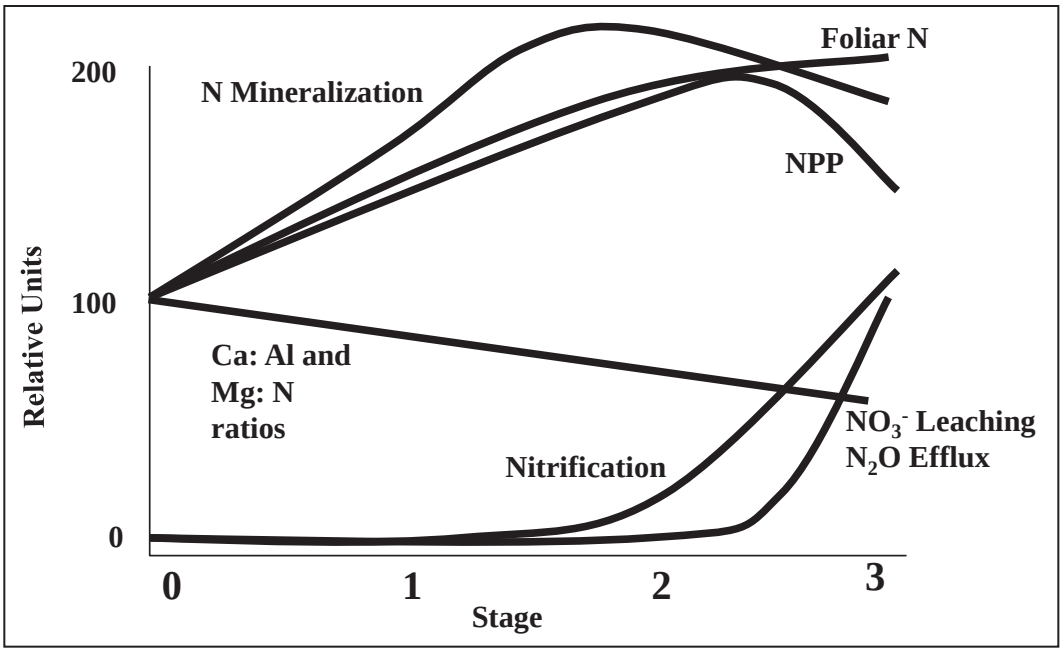

Figure 1.1—Stages of N saturation for forest ecosystems (reprinted with permission in modified format from Aber et al. 1998).

in N retention (stage 3). When vegetation is no longer N limited, but before damage is incurred, nitrate (NO_3^-) immobilization will be reduced and NO_3^- export is likely to increase gradually. The increase in NO_3^- export may occur earlier in this progression of N saturation than initially thought (Emmett 2007). Adverse consequences on plant health and growth occur in the last stage of N saturation, resulting from some combination of insufficient allocation of plant carbon to roots and mycorrhizae and soil acidification induced by NO_3^- leaching. Nutrient imbalances (e.g., elevated N:Ca or N:Mg) may also affect plant health. In addition, alterations in an ecosystem's N status may lead to increased susceptibility to secondary stresses, including freezing injury (Schaberg et al. 2002), pest outbreak, and drought (Bailey et al. 2004).

At the catchment level, N saturation is indicated by increased NO_3^- leaching, especially during the growing season (Aber et al. 2003, Stoddard 1994). Stoddard (1994) described the stages of surface water response to N saturation (Fig. 1.2).

At Stage 1, seasonal losses of NO_3^- occur during periods of hydrologic flushing (e.g., snowmelt); in stage 2, episodic NO_3^- losses remain high, and growing season losses increase as biotic demand for N is depressed; in stage 3, the base flow NO_3^- concentration becomes elevated to such an extent that seasonal patterns are no longer significant (Stoddard 1994).

Among the most significant indicators of N saturation are changes in species composition or community structure in terrestrial and aquatic ecosystems. Alterations in species composition may result from shifts in dominance of species present, for epiphytic lichens, for example (Geiser and Neitlich 2007, Jovan 2008, McCune and Geiser 2009), or increased dominance of invasive species, and may lead to significant changes in vegetation. Nitrogen deposition may affect species richness and has been implicated in dramatic declines in species richness, for example, in coastal sage scrub[1], grasslands (Stevens et al. 2004).

1.2.3 Critical Loads of N Deposition

A critical load is the level of input of a pollutant below which no harmful ecological effect occurs over the long

[1]Allen, E.B. Unpublished data. Professor and natural resources extension specialist, Department of Botany and Plant Sciences and Center for Conservation Biology, University of California, Riverside, CA 92521.

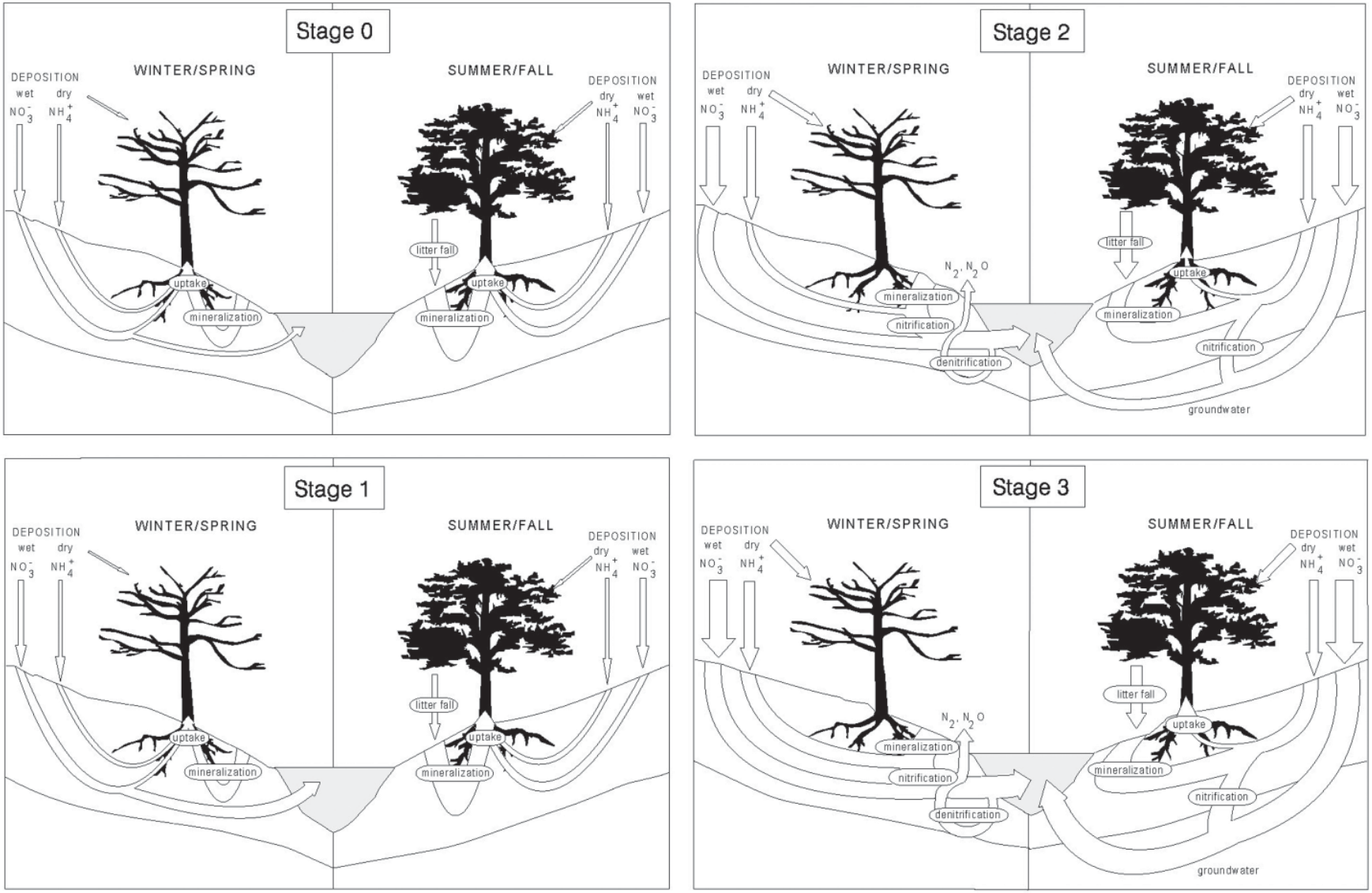

Figure 1.2—Stages of N saturation for surface water (reprinted with permission in modified format from Stoddard 1994, as published by the American Chemical Society).

term (UBA 2004). Critical loads have been defined for Europe (Posch et al. 1995, Posch et al. 2001) and have been used as a tool in the process of negotiating air pollution emission reductions in Europe. In the United States, critical loads have been calculated for specific regions (Dupont et al. 2005, Fenn et al. 2008, NEG ECP 2003, Williams and Tonnessen 2000), and are of interest to policy makers for assessing emission reduction programs and to resource managers as a tool to evaluate the potential impact of new pollution sources (Burns et al. 2008, Porter et al. 2005).

There are three main approaches for calculating critical loads: empirical, simple mass balance, and dynamic modeling approaches (Pardo 2010). Empirical approaches are based on observations of the response of an ecosystem or ecosystem component (e.g., foliage, lichens, soil) to a given, observed deposition level. Empirical critical loads are calculated using data obtained from a single site or multiple sites; 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 N based on inputs and outputs. 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 based on defining acceptable values for fluxes into and out of the ecosystem (soil N accumulation, N leaching, or acceptable/critical NO_3^- concentration in leachate).

Dynamic models use a mass balance approach expanded by incorporating internal feedbacks, such as accumulation of N in the system. Dynamic models can predict the amount of time it takes for N deposition to damage an ecosystem, as well as the amount of time necessary for recovery. Dynamic models that capture the complexity of the N cycle are currently unavailable for broad-scale use across multiple ecosystems in the United States. This review focuses on the data available to identify empirical critical loads for ecoregions of the United States.

1.3 Report Organization

This document is organized by ecoregion. The ecoregions are described in detail in Chapter 2. Issues and challenges in estimating total N deposition are described in Chapter 3. The methods are described in Chapter 4 for mycorrhizal fungi, lichens, herbaceous plants and shrubs, and forest ecosystems. For other receptors or ecosystem types, the methods are described in the individual chapters. Explanation of the response parameters and details of how the critical load was set are given in each ecoregion chapter. For each ecoregion, the ranges of responses reported are presented, and estimates for critical loads are made, where possible; the critical load values are compared to values presented for similar ecosystems types outside the United States (when possible); and the steps necessary to identify or improve critical loads estimates are described.

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