Soil Quality: Some Basic Considerations and Case Studies

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Abstract—Some fundamental properties of soils that pertain to the concept of soil quality are discussed including a discussion of what can and cannot be changed with management. Case studies showing the effects of N-fixing vegetation and N-enrichment effects on invasive species are provided to illustrate the complications that may arise from applying one soil quality standard to all cases. Finally, the "nitrogen problem is discussed: nitrogen is the most frequently growth-limiting nutrient and yet it is also the nutrient that is often the most problematic to manage without causing deleterious effects on the availability of other nutrients and water quality.

Introduction

Soil quality is a concept that, in theory, all soil scientists should embrace. Concerns have been expressed, however, that the concept is too broad and encompassing to be meaningful, and that if soil quality indicators are applied, this must be done on a site-specific basis with specific management objectives in mind (Karlen and others 1997; Page-Dumroese and others 2000; Schoenholtz and others 2000). The first questions that come to mind for this author are who will define soil quality? Soil Scientists? Farmers? Water quality experts? Conservationists? Lawyers? What criteria or outcome will be used to set the criteria for soil quality? Plant growth? Water quality? The soils themselves? Will one definition fit all? Not likely. Will the various definitions conflict? Almost certainly.

Several excellent reviews of soil quality have already been published and the reader is referred to them for details on potential criteria as well as more philosophical aspects of the issue (*e.g.*, Karlen and others1997; Page-Dumroese and others 2000; Schoenholtz and others 2000). In this paper, I will only briefly review some basic soil properties with an eye to what we might be able to change by human intervention and what cannot be, and how (if at all) these changes can be translated into the concept of soil quality.

Some Basic Soil Concepts

Factors of Soil Formation

Jenny (1941) defined factors of soil formation as parent material, climate, topography, and biota, all of which are integrated over time:

$$\text{Soil} = \int f(\text{parent material, climate, biota, topography})$$

There have been elaborations of this model over the years since its inception, but for our purposes it will suffice. The factors of soil formation that we can and often do modify include biota, most usually by modifying vegetation, and with heavy equipment and great effort we can also modify topography. Vegetation effects on soils are very well documented, including nutrient depletion by uptake (species variation being a major factor here; Binkley and Menyailo 2005). Introducing nitrogen fixing species can greatly enhance soil C and N status, but also may cause soil acidification by producing excess nitrification (Van Miegroet and Cole 1984). Further aspects of the effects of too much N fixation and other inputs of N will be discussed later. We can also modify soil biota,

Table 1. An abbreviated description of the 12 soil orders according to the U.S. 12th Approximation.

Soil order	Description
Alfisols	Clay migration, moderately high % BS
Andisols	Volcanic parent material, high P fixation
Aridisols	Arid soils, high in salts and pH
Entisols	Not well-developed even after long periods (can occur anywhere)
Gelisols	Permafrost
Histosols	Soils.formed from organic matter (peats and mucks)
Inceptisols	Still forming, water is available for soil formation
Mollisols	Organic-rich A horizons, % BS usually > 50%
Oxisols	Highly-weathered (e.g., tropical rainforest)
Spodosols	Fe, Al, and organic matter transport, whitish E Horizon (e.g., boreal forest)
Ultisols	Clay transport like Alfisols, but much more acidic; higher temperature; often highly weathered (e.g., Southeastern United States)
Vertisols	Mixed soils; swelling clays, frost, etc. cause lower horizons to mix with upper horizons; often characterized by cracks

although with far less precision, by introducing mycorrhizae (*e.g.,* Hererra and others 1993; Trappe 1997), modifying the C:N ratio either by adding N or organic C, or adding nutrients via fertilization (Miller 1981). On the other hand, we cannot modify parent material (except perhaps with heavy equipment and great difficulty), climate, or time.

The factors of soil formation listed above lead to the development of soils as we see them, and many soil classification systems have been proposed (Buol and others 2003). In the U.S. 12th approximation, (USDA 1999), the highest level classification is the soil order (table 1). The U.S. system becomes very complex and specific from that point on, and a full description of it is well beyond the scope of this paper. The 12 soil orders have some inherent properties that affect properties commonly associated with their “quality.” Some, such as Entisols and Inceptisols, can be so poorly developed because of their young age (the time factor) or parent material (for example, weathering-resistant and inherently nutrient poor quartz sand). Others, such as Mollisols and many Alfisols, can, for the same reasons, be quite rich in nutrients commonly associated with high “quality.” Many of these soil orders (for example, Ultisols, Mollisols, Oxisols, Spodosols, and especially Aridisols and Gelisols) have a rather specific geographic distribution that is influenced mainly by climate whereas others (Andisols and to some degree Vertisols) are more strongly influenced by parent material. So arises the first question regarding soil quality: do we judge soil quality from a single standard, by lumping all 12 orders and the myriad subdivisions within them into one bucket and assessing their quality from an overall standard, (which some, because of their very nature, will of course never be able to achieve), or do we restrict our assessments of soil quality to within soil order, at a minimum, and perhaps even at a lower order of classification such as great group or even lower? If the latter, we may soon find, for example, that a high “quality” Mollisol is quite a different thing from a high “quality” Oxisol.

Soil Physical Properties

The soil physical properties commonly listed in basic soil texts (Brady and Weil 2008; Gardiner and Miller 2008; Singer and Munns 2006) include texture, structure, coarse (rock) fragments (which is that particle size >2 mm by convention), bulk density, and porosity. These basic physical properties lend the soil its properties associated with water, namely, field moisture capacity (FMC) (the maximum amount of water held in the soil after drainage, typically at tensions of –10 m to –33 kPa), permanent wilting percentage (PWP) (soil water content at which plants can no longer remove water from soil, usually defined at –1,500 kPa, but for desert plants can be as high at –6,000 kPa), available water capacity (FMC-PWP), and hydraulic conductivity. Water available to plants at any given time (plant available water, PAW) is in theory equal to the difference between soil moisture content at the time in question minus soil moisture content at PWP; thus, after gravitational drainage has occurred, PAW ≤ AWC (available water

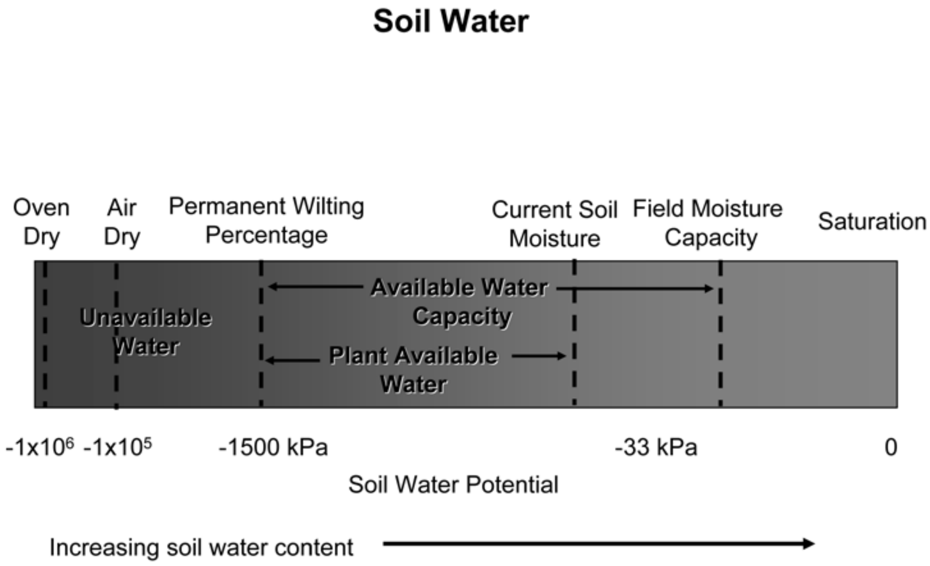


Figure 1. Schematic representation of soil water fractions.

content) (fig. 1). These soil water properties are a function of texture and structure as well as organic matter content, but this relationship is complex. The idealized (round) maximum pore size that will hold water at any given tension can be calculated from the capillary rise equation, and thus soil water properties are a function not only of total soil porosity (often calculated as 1 – bulk density/2.65 for mineral soils, assuming the density of the soil mineral fraction to be 2.65 g cm^{-2}) but also of pore size distribution, which in turn is a function of soil texture and soil structure (*e.g.,* aggregation). Thus, soils with high clay content and poor structure may have very high FMC, but will also likely have high PWP so that AWC is actually lower than soils with a loamy structure (fig. 2).

Soil structure and bulk density can both be changed by management and so can soil water properties. Soil texture cannot be changed. Adding organic matter to clay soils can improve aggregation, which in turn can lessen the problem with too many fine pores and high PWP. On the other hand, repeated disturbance leading to organic matter loss can have the opposite effect. Bulk density, total porosity, pore size distribution and, therefore, soil water properties can all be modified by management. Compaction reduces total porosity and usually creates more fine pores, perhaps increasing both FMC and PWP with variable effects on AWC. Compaction is usually seen as an undesirable effect, but, as shown by Gomez and others (1999), compaction can cause AWC and tree growth to go in either a positive or a negative direction. Specifically, Gomez and others (1999) found that compaction in clay textured soils caused the expected effect of reducing AWC, but in a sandy loam soil, compaction caused greater increases in FMC than in PWP, thus increasing AWC and tree growth. Thus, compaction can actually improve soil "quality."

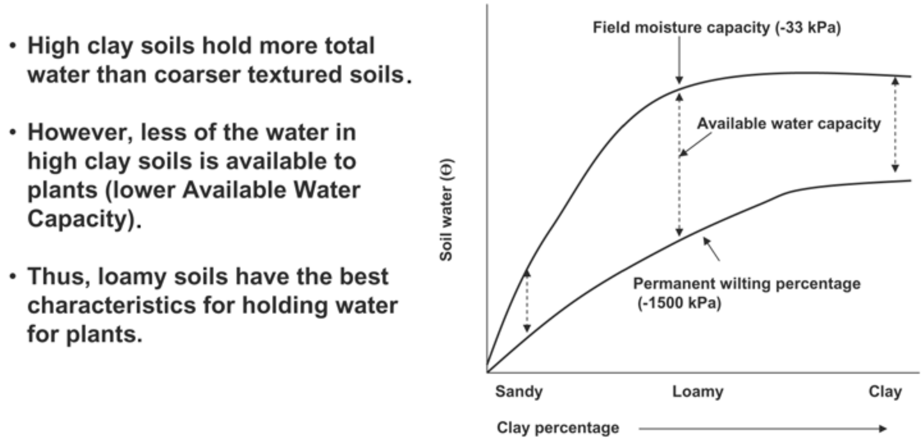

- **High clay soils hold more total water than coarser textured soils.**
- **However, less of the water in high clay soils is available to plants (lower Available Water Capacity).**
- **Thus, loamy soils have the best characteristics for holding water for plants.**

Figure 2. Schematic representation of available soil water with changing soil texture.

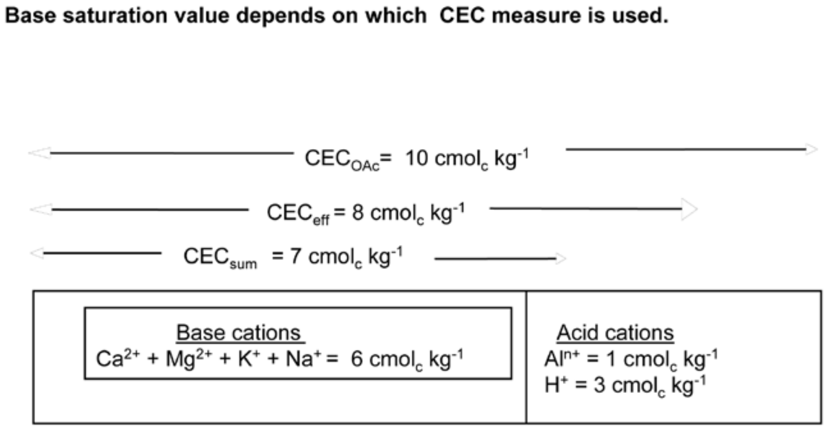
Soil Chemical Properties

Soil chemical properties commonly measured include total carbon, organic carbon, total nitrogen, C:N ratio (including organic carbon only), cation exchange capacity, exchangeable Ca^{2+}, K^+, Mg^{2+}, Na^+, Al^{n+}, base saturation, extractable NH_4^+ and NO_3^-, extractable ortho-P, extractable SO_4^{2-}, and extractable micronutrients. All of these parameters can be modified by management (*e.g.,* fertilization) and this is commonly done in agricultural and intensively managed forest soils. There are, however, problems with measurement of many of these soil chemical parameters. Total C and total N are easily measured with modern combustion equipment, but it must be borne in mind that these analyses include both organic and inorganic forms. In the case of C, this can be significant if carbonates are present, and typically they must be removed by acid pretreatment if proper C:N ratio calculations are to be made. Ammonium and NO_3^- (mineral N) analyses provide data on readily available N, analogous to “cash in the wallet”; but, like “cash in the wallet,” they do not necessarily provide a good index of soil N status as they are ephemeral quantities and turn over rapidly in soils. (To carry the analogy a step further, it could well be that the millionaire carries little cash in his wallet.) It is the flux from organic to mineral N, not the standing pool of mineral N, that gives the best estimates of N availability (as cash flow gives a better estimate of wealth than cash in the wallet), and thus there are many methods proposed to measure “N mineralization”(Robertson and others 1999). Unfortunately, all of these methods have artifacts associated with them. All N mineralization methods not involving isotopes require that plant N uptake be prevented; thus, roots must either be removed or killed. In the former case, the process of root removal may create the “assart” effect, whereby soil disturbance increases N mineralization and in the latter case, N mineralization may be artificially augmented by inputs of mineral N from decaying roots. Or, alternatively, methods that eliminate roots also eliminate the possibility of rhizosphere-enhanced soil organic matter mineralization, thus underestimating real, in-situ N mineralization (*e.g.,* Hamilton and Day 2001). Isotope methods seek to avoid these problems (see reference for a full discussion), but are expensive and have artifacts of their own to deal with. In short, there is no reliable, standard, and relatively inexpensive method for measuring N availability in soils. In this author’s opinion, organic C, organic N, and organic C:N ratio are the best compromises in terms of information provided compared to cost.

There are far fewer problems with exchangeable cations and cation exchange capacity (CEC) than there are for N, but care must be taken to be consistent with methods. Most agricultural labs in the United States use ammonium acetate for cation exchange capacity, which in theory buffers pH to 7. As noted by Sumner and Miller (1996), the continued use of this method is unfortunate because it grossly overestimates the CEC of acidic soils. The Ba-TEA method that raises pH to 8.3 causes an even greater inflation of CEC values in acid soils. While the pH in most agricultural soils is manipulated routinely, it often has a target value of 7. Seldom are forest soils at pH 7, and even more seldom do we attempt to bring them to pH 7; thus, many forest soils laboratories use the neutral salt method, (1 M NH_4Cl; Skinner and others 2001), which measures CEC at normal soil pH, or close to it, perhaps with some depression in pH due to the salt effect, where NH_4^+ displaces some exchangeable H^+ and Al^{3+} by mass action. In theory, either ammonium acetate or ammonium chloride will extract approximately the same amount of base cations, but ammonium acetate will measure a greater proportion of pH-dependent CEC, which will be an important factor in organic-rich surface horizons. Thus, the two methods could yield two different measures of base saturation for the same soil at the same time, leading the unsuspecting to believe that some change has occurred. This is illustrated schematically in figure 3, where a hypothetical soil is extracted by these two methods.

Extractable ortho-P methods yield results that are nearly as ephemeral as is the case for mineral N. We have found substantial seasonal variation in Bray-extractable P (weak solutions of HCl + NH_4F) (Johnson and Todd 1984; Johnson and others 1988) and also some inconsistent differences between Bray- and bicarbonate-extractable P (Johnson and others 1997; Susfalk 2000). Specifically, Bray extractions consistently yield greater values for P in soils derived from decomposed granite in our sites in the eastern Sierra

Figure 3. Example of how different methods for measuring cation exchange capacity can lead to misleading conclusions about base saturation.



$$\%BS_{sum} = \frac{Ca^{2+} + Mg^{2+} + K^{+} + Na^{+}}{CEC_{sum}} \quad X\ 100$$

$$= \frac{Ca^{2+} + Mg^{2+} + K^{+} + Na^{+}}{Ca^{2+} + Mg^{2+} + K^{+} + Na^{+} + Al^{n+}} \quad X\ 100 \quad = \quad \frac{6}{7} \quad 100 \; = \; \underline{85\%}$$

$$\%BS_{eff} = \frac{Ca^{2+} + Mg^{2+} + K^{+} + Na^{+}}{CEC_{rff}} \quad X\ 100 \quad = \quad \frac{6}{8} \; X\ 100 \; = \; \underline{75\%}$$

$$\%BS_{OAc} = \frac{Ca^{2+} + Mg^{2+} + K^{+} + Na^{+}}{CEC_{OAc}} \quad X\ 100 \quad = \quad \frac{6}{10} \; X\ 100 \; = \; \underline{60\%}$$

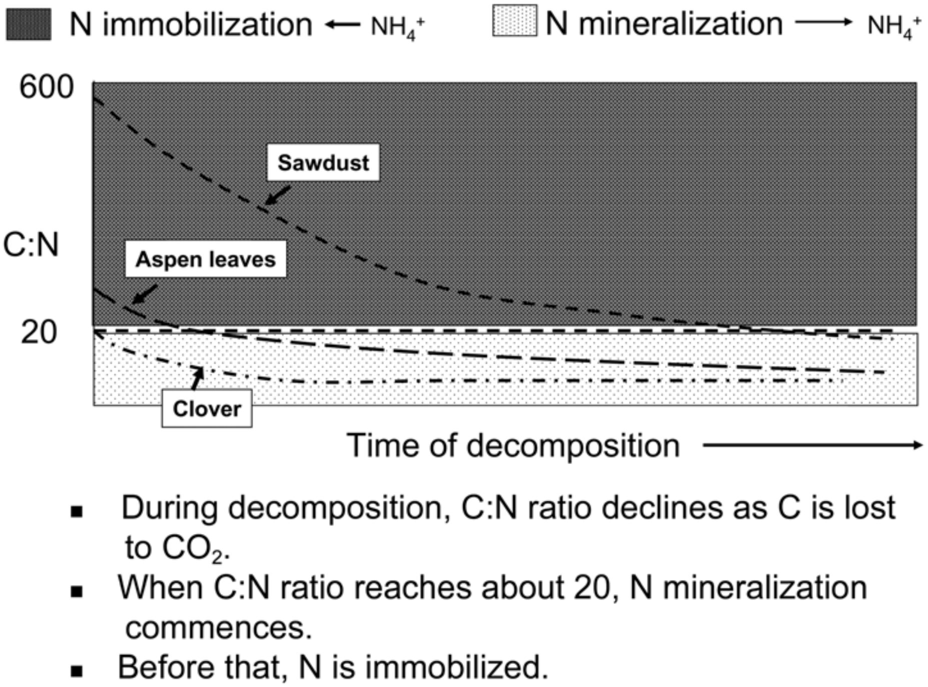
Nevada mountains (Johnson and others 2005; Susfalk 2000) whereas bicarbonate extractions often yield greater values for P in soils derived from andesite (Susfalk 2000). Susfalk (2000) studied P in soils from this region in great depth and concluded that the andesite-derived soils had far greater buffering power for ortho-P than the soils derived from decomposed granite, causing the andesitic soils to both quickly adsorb added ortho-P and also to release greater quantities of ortho-P with repeated extractions. He concluded that "a one-point P extraction index (bicarbonate-P or Bray-P) was a poor indicator of extractable P in [the andesitic soils] because it was unable to account for buffering effects, and [also a poor indicator of extractable P in granitic soils] because they extracted non-labile forms of P that may not have been plant-available."

Far less work has been done on extractable sulfate, but it is well known that pH has a very strong influence on it (Singh 1984) and that soils enriched in sesquioxides and high amounts of pollutant sulfur inputs have the highest levels of soil sulfate (Johnson 1984).

Many of the micronutrients are especially sensitive to pH. Copper, zinc, and iron, in particular, become less available at higher pH as they begin to precipitate as hydroxides. Molybdenum and, to a lesser extent, boron (being in anionic form in soil solution) become more available at higher pH as they desorb from sesquioxides. Copper is strongly absorbed by organic matter and, therefore, can be deficient in high-OM soils.

Soil Biological Properties

We can, with relative ease, cause changes in vegetation, which will in turn cause changes in soil biological and chemical properties. For example, planting with N-fixers can greatly increase soil C and N status, but also can result in greater soil acidification (Johnson and Curtis 2001; Van Miegroet and Cole 1984). We can also, to a more limited degree, change soil microbiota by introducing mycorrhizal innoculum, fertilizers, or raising C:N ratio by adding woody materials or even sugar to tie up available N. One



- During decomposition, C:N ratio declines as C is lost to CO_2.
- When C:N ratio reaches about 20, N mineralization commences.
- Before that, N is immobilized.

Figure 4. Schematic representation of the effects of soil C:N ratio on net N mineralization and immobilization.

of the easiest soil biological factors to change is that of nitrifying organisms: add available N, and they will increase their activity in most cases, probably quite quickly, and not entirely without negative consequences. We can also, with somewhat more effort, stimulate denitrifiers by creating anaerobic conditions and supplying organic substrates.

In the context of soil biology, it is perhaps germane to consider the results of long-term decomposition studies conducted by Berg and others (2003). Textbook knowledge says that adding material with high C:N ratios such as woody tissues will cause slower decomposition; also, available N immobilization as microbes, with their own C:N ratios of between 6 and 12 to 1, try to digest materials with C:N ratios an order of magnitude or more greater. In order to adjust the C:N ratios of the material they consume, they (1) release organic C as CO_2, and (2) import available N from soil pools. As a consequence, the C:N ratio of decomposing materials decreases during decomposition. As the material reaches a value of approximately 20:1, microbes begin releasing, instead of taking up, available N and, therefore, shift from the immobilization to the mineralization phase (fig. 4).

A significant complication to this textbook scenario was introduced by Berg and others (2003) with their concept of limit values. Their long-term (two-decade) decomposition studies showed that, indeed, materials with higher C:N ratios initially decomposed more slowly than those with lower C:N ratios. Over time, however, they found that materials with initially higher C:N ratios leveled off sooner, reaching a quasi-steady-state condition referred to as the limit value for remaining mass that was lower than that reached by materials with an initially lower C:N ratio. Thus, over the long term, organic C and N contents of materials with initially lower C:N ratios remained greater than those with initially higher C:N ratios—presumably this material is very humic and stable in nature, perhaps entering the stable soil organic matter pool. Thus, litter with initially greater "quality," which is often indexed as litter with lower C:N ratio, certainly leads to short-term increases in decomposition and, therefore, initially lower initial C and N pools in the O horizon. Over the long run this higher quality litter also leads to greater soil organic C and N contents (fig. 5). This example illustrates how a soil quality parameter, such as available N, can change with time—even without any intervention by humans or disturbance—simply by the nature of decomposition and its interaction with C and N in decomposing material, as manifested in both its short- and long-term effects. This reversal of N availability and C sequestration over time would seem to make an assignment of a soil quality value a very elusive thing indeed.

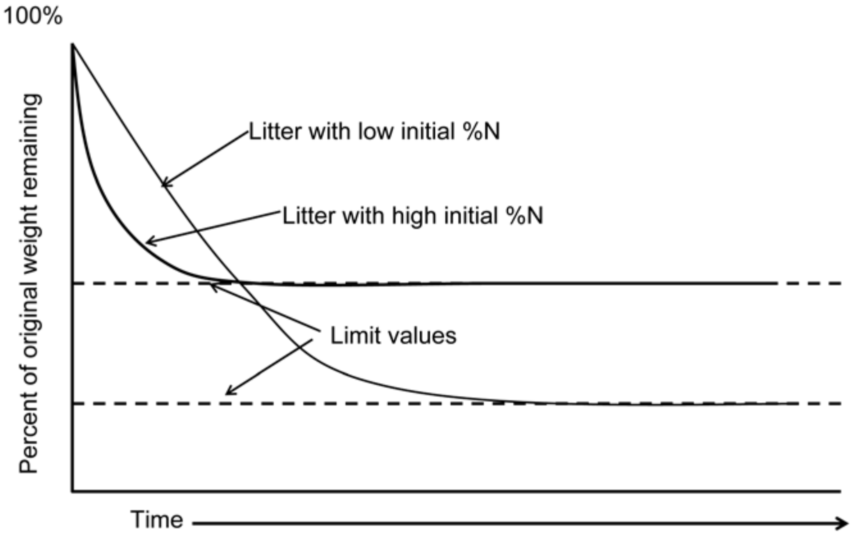


Figure 5. Schematic representation of the decomposition of litter with initially high vs. initially low C:N ratio (after Berg and others 2003).

Case Studies

So what constitutes a good quality soil? The Soil Science Society of America defined soil quality as "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation" (Karlen and others 1997). The questions being asked in the following case studies are (1) are these goals mutually compatible? (2) are these goals always desirable? and finally, and most importantly, (3) does the very concept of soil quality add to or detract from the concept of soil fertility? It seems to this author that the problem with the concept of soil quality is that it implies a blanket value judgment on soils that one fears will be applied (perhaps legally) to all soils in all situations. We are more familiar with the concept of soil fertility, and while it also implies a value judgment it is more specific (is the soil good for plant growth?) and can more readily be quantified with standard soil analyses. When we speak of water quality, we generally refer to conditions of low fertility (oligotrophic) as being desirable and waters of high fertility (eutrophic) being "polluted" and undesirable. To some degree, the same judgments may well apply to soil quality in some circumstances. For example, an endangered species that is thrifty with nitrogen is being threatened by increasing competition from a nitrogen-loving invasive species as a result of increased nitrogen fertility due to pollutant inputs. And, of course, more fertile soils are in fact more likely to produce more fertile surface waters, thus creating an automatic conflict between the concepts of soil and water quality.

Case Study 1: The Effects of Red Alder on Soil Quality

Red alder (*Alnus rubra*) is a native N-fixing tree in the Pacific Northwest that occupies sites after disturbances such as fire, logging, erosion, etc. It has long been known that red alder improves soil N status and that subsequent stands of Douglas-fir (*Pseudotsuga menziesii*) greatly benefit from this (Binkley 2003). A little over two decades ago, however, Van Miegroet and Cole (1984) found that red alder simply cannot stop itself from fixing N even when no more N is needed by any biological entity in the ecosystem except nitrifying bacteria. Nitrification results in the creation of nitrate and acid, and when the nitrate is not taken up, the acid exchanges for base cations on the soil exchange complex, nitrate leaches and soil acidifies. There is also the suggestion that soil available P is tied up because of the excess N, but this is somewhat more controversial (Compton and others 1998; Giardina and others 1998). Furthermore, red alder apparently makes soil conditions less suitable for itself, and does not do well on sites formerly occupied by red alder (Van Miegroet and others 1992). Douglas-fir, on the other hand, is not bothered by the acidity or lower soil P status and thrives in the N-rich environment. How, then, does one define soil quality in this instance? The quality for red alder is poor, that for Douglas fir is good, and, while red alder occupies the site, water quality is degraded by high nitrate concentrations (which cease quickly after the red alder is cut down, apparently causing cessation of N fixation). Compton and others (2003) find that the degree of occupancy of red alder in watersheds was directly related to the concentrations of nitrate in streamwaters.

Case Study 2: The Effects of Snowbrush on Soil Fertility

Snowbrush (*Ceanothus velutinus*) is a major fire-adapted N-fixing species occurring on disturbed sites in the Sierra Nevada and southern Cascade mountains. The heat generated from fire is thought to be the major mechanism for seed germination (Gratkowski 1962). Nitrogen (N) fixation in snowbrush is associated with *Frankia* spp. actinomycetes, and reported fixation rates of up to 142 kg N ha^{-1} yr^{-1} (Binkley and others 1982; Youngberg and Wollum 1976). Snowbrush is generally intolerant to shade, but it has been observed to dominate a site for over 50 years following a wildfire by suppressing forest regeneration (Conrad and Radosevich 1982). Zavitkovski and Newton (1968) describe four stages to snowbrush growth following fire or disturbance: (1) growth with little or no accumulation of organic matter between the ages of 1 to 5 years; (2) rapid growth with increasing biomass, increase of N in biomass, and accumulation of organic matter between the ages of 5 to 10 years; (3) an equilibrium stage, which lasts between the ages of 10 to 15 or as long as 50 years; and (4) the final stage of decline and decomposition, usually due to the growth of a forest canopy shading the snowbrush out.

We have made several comparisons of soils beneath snowbrush and adjacent Jeffrey pine (*Pinus jeffreyi*) stands in Little Valley, Nevada, a site just east of Lake Tahoe (Johnson 1995; Johnson and others 2005). We have found that soils beneath snowbrush consistently have lower bulk density and higher total C and total N concentrations than in adjacent pine stands, as would be expected. Unlike the case with red alder, however, we have also found that there are no differences in extractable P concentrations in soils beneath snowbrush as compared to soils beneath pine, but snowbrush soils have consistently higher concentrations of exchangeable Ca^{2+}, K^{+}, and Mg^{2+} than in adjacent pine stands. The higher base cation status beneath snowbrush is consistent with the fact that we have found no evidence of elevated levels of NO_3^- leaching beneath snowbrush stands (Johnson 1995; Stein 2006).

Thus, it appears that snowbrush improves soil bulk density and nutrition—soil "quality"—in most measurable ways. Is this a desirable outcome? Purely from the perspective of soil "quality" it certainly is; however, because snowbrush does not "poison itself out" of a site like red alder apparently does, it can persist for many decades after fire and prevent reforestation. Thus, we have high quality soils but an undesirable vegetative cover to go with these high quality soils.

Case Study 3: Cheatgrass in the Great Basin

Cheatgrass (*Bromus tectorum*), an exotic annual grass, is rapidly expanding throughout the Great Basin. This highly competitive invader is resulting in the widespread deterioration of mid- to low-elevation sagebrush ecosystems and, more recently, salt desert ecosystems (Brooks and Pyke 1991). Cheatgrass has altered fire regimes in native ecosystems because it increases fine fuels, is highly flammable, and increases the rate of fire spread (Link and others 2006). In many parts of the region an annual grass-fire cycle now exists in which fire return intervals have decreased from about 60 to 110 years to as little as 3 to 5 years (Whisenant 1990). Recent field studies have shown the importance of available inorganic nitrogen in controlling cheatgrass establishment and growth (McLendon and Redente 1991; Young and others 1999). Experiments with sugaring soils to stimulate microbial competition for N, thus reducing mineral N supplies in soils have proven to severely limit cheatgrass growth and to favor native species by reducing competition (Young and others 1999).

An alternative approach to sugaring to tie up mineral N might be to reduce total N supplies and, therefore, mineral N supplies by repeated burning. It is well documented that nearly all N contained in organic material that is burned is volatilized and lost from the system, potentially causing long-term declines in ecosystem N capital unless the N is replaced by atmospheric deposition, N-fixation, or fertilization (Blair 1997; Neary and others 1999; Raison and others 1985). On the other hand, burning commonly causes short-term increases in soil ammonium levels because of the heat-induced denaturing of soil organic N (Neary and others 1999). The pulse of ammonium is often followed by a pulse of nitrate and nitrate leaching once nitrifying bacteria occupy the site again. The short-term pulse of ammonium after fire is thought to be one factor favoring nitrophilic cheatgrass after rangeland fire (Monaco and others 2003). Over the long-term, however, one would expect that repeated burning without replacement of lost N could cause reductions in soil mineral N levels, at least after the initial post-fire pulse has passed (Blair 1997; Johnson and Matchett 2001; Ojima and others 1994). This is attributed to both volatile losses of N and also to a form of progressive N deficiency, where N concentrations in vegetation decline over time in response to reduced soil N availability, causing inputs of detritus with lowering C:N ratios. The N deficiency is further exacerbated by increasing microbial competition for N in much the same manner as the short-term sugaring experiments described above (Blair 1997; Johnson and Matchett 2001; Ojima and others 1994). Although it has been shown that cheatgrass invasion can rapidly alter N cycling (Evans and others 2001), little is known about the effects of repeated fire on N availability in cheatgrass dominated rangelands. Our objective in this study is to explore the prospects for "burning out" cheatgrass with repeated fires designed to reduce total and available soil N and, consequently, cheatgrass growth and reproduction.

The preceding discussion clearly shows that greater soil "quality"—specifically, better soil N status—favors one of the most destructive invasive species in the Great Basin. Reducing soil N availability (quality?) seems to be the best, albeit somewhat faint, hope of controlling this species.

The Nitrogen Problem

Nitrogen is unique among nutrients in many ways. Unlike P, K, Ca, and Mg, for example, N is rarely present in parent rock, and in the vast majority of cases is naturally introduced to the soil from the atmosphere by symbiotic and non-symbiotic fixation, atmospheric deposition, and lightning. The major inorganic forms of N in soils include both a cation (NH_4^+) that is strongly absorbed to soils and an anion (NO_3^-) that is very weakly absorbed to soils. Unlike most other macronutrients, however, inorganic forms of nitrogen do not persist in non-aridic soils for long. Nitrogen is the most frequently limiting nutrient in terrestrial ecosystems, and inorganic forms of N are rapidly taken up and depleted under N deficient conditions that are common in terrestrial ecosystems. When N is supplied in excess of biological demand, it does not accumulate to any significant degree on soil exchange sites or as precipitation. While the NH_4^+ form is strongly held on soil exchange sites, it does not persist for long in most soils, even when N supplies greatly exceed biological demand, before it is converted to NO_3^- during the nitrification process (which also produces H^+ - nitric acid). Nitrate so produced, if not taken up, is poorly adsorbed and will leach from soils, taking with it base cations and thus acidifying the soil. In contrast, additions of P, K, Ca, or Mg in excess of biological demands can result in large and prolonged accumulations of the ionic forms of these nutrients on exchange and adsorption sites.

Thus, N is a difficult nutrient to manage, which is unfortunate because it is the most often limiting nutrient. All nutrients (in fact all substances, including water) have regions of deficiency, sufficiency, and toxicity with increasing supplies (fig. 6). We can think of this classical curve not only in terms of plant response, but also environmental response. So, for example, while the sufficiency plateau for K, Ca, and Mg can be quite broad, as supplies of these nutrients exceed biological demands and accumulate on soil exchange or adsorption sites, the sufficiency plateau for N is very narrow, with little space between deficiency on the left and undesirable consequences such as excessive

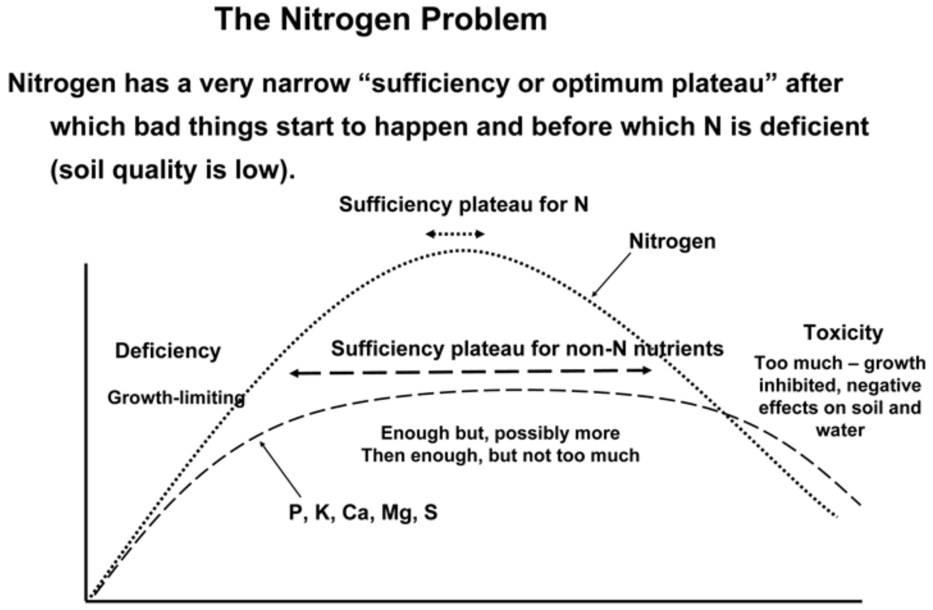


Figure 6. Schematic representation of deficiency-sufficiency-toxicity vs. nutrient supply curves for nitrogen and other nutrients.

nitrate leaching on the right. In the cases of K, Ca, and Mg, increases in leaching rates may occur without undue harm to water quality; in the case of P, leaching is nearly always minimized by various adsorption and precipitation processes in the soil and most P transport from the ecosystem is by erosion. For N, on the other hand, there is a very narrow sufficiency plateau for N between the deficiency region and the point quickly thereafter when excess N leaches from the system and degrades water quality. Thus, trying to manage for a high quality soil will necessarily require that it have good N status, but it is very difficult to achieve good N status without tipping over the edge into the toxicity region.

References

Berg, Bjorn A.; De Santo, V.; Rutigliano, F.A.; Fierro, A.; Ekbohm, G. 2003. Limit values for plant litter decomposing in two contrasting soils—influence of litter elemental composition. Acta Oecologia 24: 295-302.

Binkley, Dan. 2003. Seven decades of stand development in mixed and pure stands of conifers and nitrogen-fixing red alder. Can. J. For. Res. 33: 2274-2279.

Binkley, Dan; Cromack, K., Jr.; Fredriksen, R.L. 1982. Nitrogen accretion and availability in some snowbush ecosystems. Forest Sci. 28: 720-724.

Binkley, Dan; Menyailo, O., eds. 2005. Tree Species Effects on Soils: Implications for Global Change. NATO Science Series. Springer, Dordrecht.

Blair, John M. 1997. Fire, N availability, and plant response in grasslands: a test of the transient maxima hypothesis. Ecology 78: 2359-2368.

Brady, Nyle C.; Weil, R.R., eds. 2008. The Nature and Properties of Soils. Fourteenth Edition. Prentice Hall, Upper Saddle River, New Jersey. 965 p.

Brooks, M.L.; Pyke, D.A. 2001. Invasive plants and fire in the deserts of North America. Pages 1-14 in K.E.M. Galley, T.P. Wilson, eds. Proceedings of the Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: the First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous Publication No. 11, Tall Timbers Research Station, Tallahassee, Florida.

Buol, S.W.; Southard, R.J.; Graham, R.C.; McDaniel, P.A. 2003. Soil Genesis and Classification. Fifth Edition. Blackwell Publishing.

Compton, Jana E.; Cole, D.W. 1998. Phosphorus cycling and soil P fractions in Douglas-fir and red alder stands. For. Ecol. Manage. 110: 101-112.

Compton, Jana E.; Robbins Church, M.; Larned S.T.; Hogsett, W.E. 2003. Nitrogen export from forested watersheds in the Oregon Coast Range: the role of N2-fixing red alder. Ecosystems. 6: 773-785.

Conrad, Susan G.; Radosevich, S.R. 1982. Growth responses of white fir to decreased shading and root competition by montane chaparral shrubs. Forest Science. 28: 309-320.

Evans, R.D.; Rimer, R.; Sperry, L.; Belnap, J. 2001. Exotic plant invasion alters nitrogen dynamics in an arid grassland. Ecological Applications. 11: 1301-1310.

Gardiner, D.T.; Miller, R.W. 2008. Soils in our environment. Eleventh Edition. Pearson/Prentice Hall, Upper Saddle Hill, New Jersey. 600 p.

Giardina, Christian; Huffman, S.; Binkley, D.; Caldwell, B. 1995. Alders increase phosphorus supply in a Douglas-fir plantation. Can. J. For. Res. 25: 1652-1657.

Gomez, A.; R. Powers, R.F.; Singer, M.J.; Horwath, W.R. 2002. Soil compaction effects on growth of young ponderosa pine following litter removal in California's Sierra Nevada. Soil Sci. Soc. Am. J. 66: 1334-1343.

Gratkowski, J.H. 1962. Heat as a factor in germination of seeds of *Ceanothus velutinus* var. *laevigatus* T. and G. Dissertation, Oregon State University, Corvallis.

Hamilton, E.W.; Frank, D.A. 2001. Can plants stimulate soil microbes and their own nutrient supply? Evidence from a grazing tolerant grass. Ecology. 82(9): 2397-2402.

Herrera, M.A.; Salamanca, C.P.; Barea, J.M. 1993. Inoculation of woody legumes with selected arbuscular mycorrhizal fungi and rhizobia to recover desertified Mediterranean ecosystems Appl. Environ. Microbiol. 59(1): 129-133.

Jenny, Hans. 1941. Factors of Soil Formation. A System of Quantitative Pedology. McGrawHill Book, New York.

Johnson, Dale W. 1984. Sulfur cycling in forests. Biogeochemistry. 1: 29-44.

Johnson, Dale W. 1995. Soil properties beneath *Ceanothus* and pine stand in the eastern Sierra Nevada. Soil Sci. Soc. Am. J. 59: 918-924.

Johnson, Dale W.; Curtis, P.S. 2001. Effects of forest management on soil carbon and nitrogen storage: Meta Analysis. For. Ecol. Manage. 140: 227-238.

Johnson, Dale W.; Todd, D.E. 1984. Effects of acid irrigation CO_2 evolution, extractable nitrogen, phosphorus and aluminum in a deciduous forest soil. Soil Sci. Soc. Am. J. 48: 664-666.

Johnson, Dale W.; Henderson, G.S.; Todd, D.E. 1988. Changes in nutrient distribution in forests and soils of Walker Branch Watershed, Tennessee, over an eleven-year period. Biogeochemistry. 5: 275-293.

Johnson, Dale W.; Murphy, J.D.; Susfalk, R.B.; Caldwell, T.G.; Miller, W.W.; Walker, R.F.; Powers, R.F. 2005. The effects of wildfire, salvage logging, and post-fire N fixation on the nutrient budgets of a Sierran forest. For Ecol. Manage. 220: 155-165.

Johnson, Dale W.; Susfalk, R.B.; Dahlgren, R.A. 1997. Nutrient fluxes in forests of the eastern Sierra Nevada Mountains, USA. Global Biogeochem. Cycles. 11: 673-681.

Johnson, L.D.; Matchett, J.R. 2001. Fire and grazing regulate belowground processes in tallgrass prairie. Ecology. 82: 3377-3389.

Karlen, D.L.; Mausbach, M.J.; Doran, J.W.; Cline, R.G.; Harris, R.F.; Schuman. G.E. 1997. Soil quality: a concept, definition, and framework for evaluation. Soil Sci. Soc. Am. J. 61: 4-10.

Link, S.O.; Keeler, C.W.; Hill, R.W.; Hagen, E. 2006. *Bromus tectorum* cover mapping and fire risk. International Journal of Wildland Fire. 15: 113-119.

McLendon, T.; Redente, R.F. 1991. Nitrogen and phosphorus effects on secondary succession dynamics on a semi-arid sagebrush site. Ecology. 72: 2016-2024.

Miller, Hugh G. 1981. Forest fertilization: some guiding concepts. Forestry. 54: 157-167.

Monaco, T.A.; Johnson, D.A.; Norton, J.M.; Jones, T.A.; Connors, K.J.; Norton, J.B.; Redinbaugh, M.B. 2003. Contrasting responses of Intermountain West grasses to soil nitrogen. Journal of Range Management. 56: 289-290.

Neary, Daniel G.; Klopatek, C.C.; DeBano, L.F.; Ffolliot, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. For. Ecol. Manage. 122: 51-71.

Ojima, Dennis S.; Schimel, D.W.; Parton, W.J.; Owensby, D.W. 1994. Long- and short-term effects of fire on nitrogen cycling in a tallgrass prairie. Biogeochemistry. 24: 67-84.

Page-Dumroese, Deborah; Jurgensen, M.; Elliot, W.; Rice, T.; Nesser, J.; Collins, T.; Meurisse, R. 2000. Soil quality standards and guidelines for forest sustainability in northwestern North America. For. Ecol. Manage. 138: 445-462.

Raison, R. John; Khanna, P.K.; Woods, P.V. 1985. Mechanisms of element transfer to the atmosphere during vegetation fires. Can. J. For. Res. 15:132-140.

Robertson, G. Phillip; Wedin, D.; Groffman, P.M.; Blair, J.M.; Holland, E.A.; Nadelhoffer, K.J.; Harris, D. 1999. Soil carbon and nitrogen availability: nitrogen mineralization, nitrification, and soil respiration potentials. Pages 258-271 in G.P. Robertson, D.C. Coleman, C.S. Bledsoe, and P. Sollins, eds. Standard soil methods for long-term ecological research. Oxford University Press, New York. 462 p.

Schoenholtz, Stephen H.; Van Miegroet, H.; Burger, J.A. 2000. A review of chemical and physical properties as indicators of forest soil quality: challenges and opportunities. For. Ecol. Manage. 138: 335-356.

Singer, Michael J.; Munns, D.N. 2006. Soils: An Introduction. Pearson/Prentice Hall, Upper Saddle Hill, New Jersey, USA. 466 p.

Singh, Ball R. 1984. Sulfate sorption by acid forest soils: 3. Desorption of sulfate from adsorbed surfaces as a function time, desorbing ion, pH and amount of adsorption, Soil Sci. 138: 189-197.

Skinner, Malcolm F.; Zabowski, D.; Harrison, R.; Lowe, A.; Xue, D. 2001. Measuring the cation exchange capacity of forest soils. Commun. Soil Sci. Plant Anal. 32: 1751-1764.

Stein, Chad M. 2006. Post-fire vegetation in Sierran forests: effects on nitrogen fixation, soil nitrogen availability, and water quality. MS thesis. University of Nevada, Reno.

Sumner, M.E.; Miller, W.P. 1996. Cation exchange capacity and exchange coefficients. Ch. 40. in: Sparks, D.W.; A.L. Page; P.A. Helmke; R.H. Loeppert;, P.N. Soltanpour; M.A. Tabatabai; C.T. Johnson; and M.E. Sumner. Methods of Soil Analysis. Part 3. Chemical Methods. Soil Science Society of America, Madison, WI.

Susfalk, Richard B. 2000. Relationships of soil-extractable and plant-available phosphorus in forest soils of the eastern Sierra Nevada. MS thesis. University of Nevada, Reno. 227 p.

Trappe, James M. 1977. Selection of fungi for ectomycorrhizal inoculation in nurseries. Annual Review of Phytopathology. 15: 203-222.

Van Miegroet, Helga; Cole, D.W. 1984. The impact of nitrification on soil acidification and cation leaching in a red alder ecosystem. J. Environ. Qual. 13: 586-590.

Van Miegroet, Helga; Homann, P.S.; Cole, D.W. 1992. Soil nitrogen dynamics following harvesting and conversion of red alder and Douglas-fir stands. Soil Sci. Soc. of Am. J. 56: 1311-1318.

Whisenant, S.G. 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. in E.D. McArthur; E.M. Romney; S.D. Smith; P.T. Tueller, comps. Proceedings—Symposium on cheatgrass invasion, shrub die-off and other aspects of shrub biology and management. Gen. Tech. Rep. INT-276, Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

Young, J.A.; Allen, F.A. 1996. Cheatgrass and range science: 1930-1950. J. Range Manage. 50: 530-535.

Young, James A.; Blank, R.R.; Clements, D.C. 1999. Nitrogen enrichment and immobilization influences on the dynamics of annual grass community. Pages 279-281 in David Eldridge; and David Freudenberger, eds. People and rangelands building the future. Proceedings of the VI International Rangeland Congress. Townsville, Queensland, Australia; July 19-23.

Youngberg, Chester T.; Wollum, A.G., II. 1976. Nitrogen accretion in developing *Ceanothus velutinus* stands. Soil Sci. Soc. Am. J. 40: 109-112.

Zavitkovski, J.; Newton, M. 1968. Ecological importance of snowbrush, *Ceanothus velutinus*, in the Oregon Cascades. Ecology. 49: 1134-1145.
