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Toward Improved Water Quality in Forestry: Opportunities and Challenges in a Changing Regulatory Environment

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Environmental resource managers have long recognized the importance of protecting water quality to safeguard human health and ecological resources. In North America, the Soil Conservation District movement, initiated in 1935 with passage of the Soil Conservation Act, represents early interest in this goal (Ice 2004). The 1948 Federal Water Pollution Control Act (Ch. 758; P.L. 845) and subsequent amendments (e.g., the 1972 Federal Water Pollution Control Act or "Clean Water Act" [CWA] and amendments) authorized other federal, state, and local entities to prepare programs that would reduce pollution of interstate waters and tributaries and improve the condition of surface and underground waters.[1]

The CWA makes a distinction between point and nonpoint sources. "The term *point source* means any discernible, confined and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel or other floating craft, from which pollutants are or may be discharged. This term does not include agricultural stormwater discharges and return flows from irrigated agriculture" (33 USC § 1362). Point sources are subject to regulation and control through permit requirements of the National Pollution Discharge Elimination System (NPDES) (33 USC § 1342).

"Nonpoint sources" (NPS) of water pollution are those that do not meet the definition of a point source. The US Environmental Protection Agency's (EPA) NPS web page states that NPS pollution "comes from many diffuse sources" and "generally results from land runoff, precipitation, atmospheric deposition, drainage, seepage or hydrologic modification."[2] The CWA directs the states to establish NPS pollution management programs and, with technical assistance from the EPA, to develop and implement best management practices (BMPs) for reducing pollutant loadings from NPS (33 USC § 1329).

The EPA has defined most forestry activities as NPS [40 CFR § 122.27(b)(1)], and all states with significant levels of forest management activities have developed and implemented forestry NPS control programs (National Council for Air, and Stream Improvement, Inc. [NCASI] 2009). BMPs are a primary approach in these programs and have been defined in this context as "a practice or usually a combination of practices that are determined by a state or a designated planning agency to be the most effective and practicable means (including technological, economical, and institutional considerations) of controlling point and nonpoint source pollutants at levels compatible with environmental quality goals—*note* BMPs were conceptualized in the 1972 US Federal Water Pollution Control Act" (Society of American Foresters 2010) Examples of forestry BMPs include establishing of streamside management zones, planning of road systems and harvest operations to minimize the number of stream crossings, managing surface runoff from road systems by directing it away from streams, and developing handling and application procedures for fertilizers and pesticides that prevent spills and direct application into waterbodies.

State BMP programs have been very successful in controlling the impacts of forestry operations on water quality (Archey 2004, Ice et al. 2004a). A large and growing body of research shows that forestry BMPs reduce water quality impacts by 80–90% compared with impacts from historic practices (Ice 2004, McBroom et al. 2008, NCASI 2012, Sugden et al. 2012) and provide substantial protection to maintain water quality and aquatic habitat (Williams et al. 2000, Vowell and Frydenborg 2004, Wilkerson et al. 2010). For example, vegetation retained in streamside management zones (SMZs) takes up nutrients (immobilization) in subsurface runoff and impedes movement of surface runoff, thus enhancing deposition of sediments (and associated nutrients and organic matter) within the SMZs (Walbridge 1993). SMZs have the ability to store and transform soluble inorganic nutrient forms through adsorption of phosphorus (P) and denitrification of nitrogen (N) (Walbridge and Lockaby 1994). Nationwide, it is estimated that forestry BMPs are applied nearly 90% of the time (Ice et al. 2010). Although the use of BMPs is in some sense voluntary, participation in certification programs is contingent on applying BMPs, and in many locations, they are incorporated into state regulations, as is evident from the high implementation rates nationwide.

Improvement in forestry NPS control programs is attributable to the ongoing efforts of diverse parties including the EPA, state forestry and environmental agencies, forest owners and operators, wood procurement organizations, research organizations, and environmental groups. In recent years, forest certification programs have placed additional emphasis on compliance with state BMPs and other water quality regulations (Simpson et al. 2008). An important indicator of success is the fact that forestry is no longer identified by the EPA as among the 10 leading sources of water quality impairment in the United States (US EPA 2011). In most states, the impacts of forestry on water resources are

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minor relative to the impacts attributed to other NPS categories.[3] Indeed, the quality of water draining from forested watersheds is the best in the nation relative to that for other land uses, whether the forests are managed or left untouched (Binkley and Brown 1993, Schoonover et al. 2006, Argerich et al. 2013). Thus, although clearly there remain problems such as legacy roads, difficulties with unstable soil types, and other issues, forestry BMPs are generally applied and largely successful for improving water quality.

The success of forestry BMP programs notwithstanding, there is a clear trend toward greater emphasis on fixed, numeric water quality standards in forestry regulations. This may reflect a desire for certainty of outcomes (i.e., a perception that not every forestry operation is done properly). We base this assessment on the following observations:

- Nine of the 10 leading sources of water quality impairment in the United States are NPS (US EPA 2011). Thus, Congress, the EPA, and others are searching for ways to make NPS more accountable for nonattainment of water quality standards.[4] Forestry activities can be caught up in these initiatives even if they are not major or substantial contributors to water quality impairment.
- The CWA established a Total Maximum Daily Load (TMDL) program that identifies point and NPS contributing to nonattainment of water quality standards in specific watersheds (33 USC § 1313; 40 CFR § 130.7). Some states are implementing the TMDL program in ways that can lead to new regulatory constraints and monitoring requirements on NPS, including forestry operations (California Environmental Protection Agency, North Coast Regional Water Quality Control Board 2001, Oregon Department of Environmental Quality 2011).
- Recent decisions by federal courts (*Northwest Environmental Defense Center v. Brown*,[5] *National Cotton Council v. Environmental Protection Agency*[6]) have identified traditional forestry NPS activities (roads and herbicide applications, respectively, in the two rulings) as point sources subject to permitting requirements of the CWA's NPDES. These requirements include water quality-based effluent limitations, i.e., permit conditions designed to ensure that discharges from point sources do not cause violations of water quality standards.

The implications of using water quality standards to regulate forestry depend in part on attributes of the standards themselves. Components of standards include designated uses, criteria designed to protect designated uses, and antidegradation policies. Criteria may be numeric or narrative (qualitative), with current policy requiring numeric criteria when necessary to protect designated uses. A recent EPA memorandum regarding stormwater regulations (Hanlon 2010, p. 3) recommended that "NPDES permitting authorities use numeric effluent limitations where feasible as these types of effluent limitations create objective and accountable means for controlling stormwater discharges." Thus, many states are developing numeric water quality criteria, with uncertain implications regarding NPS activities such as forestry. About one-half of states have completed or provided the EPA with an estimated date for completing some phase of criteria development for total N or total P for rivers and streams.[7]

Forest Water Quality Issues

Although setting numeric criteria seems like a logical step toward the laudable goal of achieving cleaner water, there may be challenges to applying those criteria if, for example, a cause of impairment cannot be determined. In this article, we explore this and other challenges associated with application of numeric criteria to four important aspects of water quality that can potentially be affected by forestry practices: nutrients, sediment, dissolved oxygen, and temperature.

Nutrients

Nutrients, principally N and P, are essential for aquatic communities, but excess amounts can impair water quality by promoting algal blooms and causing other adverse impacts on stream and lake biota (Binkley et al. 1999, 2004). Today there is little debate over the need to control nutrient discharges to surface waters in certain situations, and for more than a decade the EPA and most states have been attempting to develop biologically relevant numeric criteria for nutrients (e.g., Florida Department of Environmental Protection 2012).

In establishment of state water quality standards for nutrients, an important question is, Are numeric nutrient criteria adopted as state regulations useful in managing nutrient pollution? To be useful, nutrient criteria should provide an indication, within an acceptable degree of uncertainty, of whether designated uses of a waterbody are being attained. Two fundamental challenges in establishing effective numeric nutrient criteria are determining whether beneficial uses are impaired or unimpaired and establishing a quantitative relationship between a measure of impairment and a measure of nutrient availability (McLaughlin 2011).

The EPA encountered these challenges when developing a proposed numeric criterion for total phosphorus (TP) for colored lakes (i.e., those with high natural organic contents) in Florida (US EPA 2009). The agency selected chlorophyll *a* (Chl *a*) as a metric of beneficial use and defined 20 μg/L Chl *a* as the impairment threshold. The agency then used a relationship between Chl *a* and TP for colored lakes to establish a TP criterion of 0.05 mg/L (Figure 1). To highlight issues associated with uncertainty, we have added to Figure 1 a horizontal line at the 20 μg/L Chl *a* impairment threshold and a vertical line at the 0.05 mg/L TP criterion, thus dividing the figure into four quadrants. The lower left quadrant $(1 - \alpha)$ contains Chl *a* and TP measurements from colored lakes in which values for both variables are below their respective threshold/criterion, indicating beneficial use attainment. The upper right quadrant $(1 - \beta)$ contains Chl *a* and TP measurements in which values for both variables are above their respective threshold/criterion, indicating nonattainment. In the lower right, α quadrant, the incorrect decision represents a "false rejection" error because Chl *a* levels do not exceed the proposed impairment threshold. Conversely, the upper left β quadrant represents a "false acceptance" error in which Chl *a* is actually impaired even though the TP comparison indicated that it is not (Table 1). In this example, the value of the TP criterion directly affects the distribution of incorrect predictions between the upper left and lower right quadrants in Figure 1. There are many sites where TP limits are exceeded, but the water body is not impaired, indicating an overprotective criterion that has high potential to impose costly controls on TP in watersheds where Chl *a* is already below its beneficial use threshold. This problem arises because factors other than TP also influence Chl *a*, thus making the relationship statistically noisy rather than precise, as fixed regulatory limits would seem to imply.

In response to such problems, Florida has proposed site-specific alternative numeric nutrient criteria protocols that allow a higher total nitrogen or TP criterion if sufficient Chl *a* data are available to show attainment of a specific Chl *a* level that "protects against an imbalance in the natural populations of aquatic flora or fauna" [62–302.531(2)(a), F.A.C.]. This alternative criterion approach reduces the false-positive

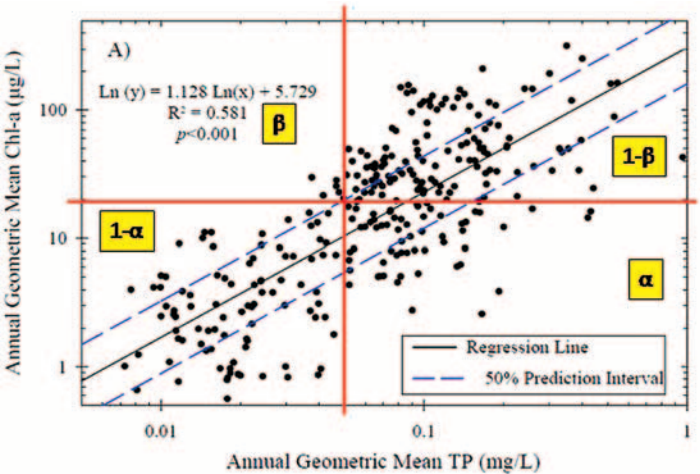

Figure 1. Relationship between Chl *a* and TP concentration for Florida's colored lakes. The horizontal line at 20 μg/L Chl *a* was added to show the proposed Chl *a* criterion for Florida's colored lakes; a vertical line at the proposed TP baseline criterion of 0.05 mg/L is also shown, along with the resulting four decision framework quadrants. (Data from US EPA 2009, McLaughlin 2011, with permission; Figure 1-11, panel A.)

Table 1. Interpretation of four decision framework quadrants shown in Figure 1.

Quadrant	Quadrant name	Indicates
Lower left	$1 - \alpha$ quadrant	TP and Chl *a* agree: both criteria attained, waterbody not impaired for Chl *a*
Upper right	$1 - \beta$ or "power" quadrant	TP and Chl *a* agree: criteria not attained, waterbody impaired for Chl *a*
Lower right	α quadrant	TP and Chl *a* do not agree: TP indicates "impaired" when Chl *a* $\leq$20 μg/L
Upper left	β quadrant	TP and Chl *a* do not agree: TP indicates "unimpaired" when Chl *a* >20 μg/L

Statistical "power" is defined as $1 - \beta$ and is the likelihood of correctly rejecting the null hypothesis.

finding rate for waters with elevated nutrients but no algal impairment.

It is clear that many factors influence concentrations and forms of nutrients in waterbodies and their effects on aquatic ecosystems. These factors differ across the United States and even within a single basin (Smith et al. 2003) and present challenges to application of numeric criteria for nutrients to forested landscapes. For example, numeric criteria based on total loadings of N and P may not reflect the actual risk of these nutrients due to low bioavailability of the organic forms of N and P (Binkley et al. 2004). Nutrients can be added to forest streams from many natural sources such as alder forests (Compton et al. 2003), weathering, and soil formation (Holloway et al. 1998), leaf litter inputs (Eshleman et al. 1998), atmospheric deposition (Lovett 1994), and returning salmon (Bilby et al. 1996). Seasonal variation in processes controlling nutrient availability and transport in watersheds can have substantial effects on stream nutrient concentrations (Mulholland 1992). For example, Carpenter (2003) reported that some stream reaches in the Clackamas River basin in Oregon may accumulate nutrients in algae during the summer and release the nutrients downstream with autumn dieback. Finally, legacy effects due to past use of the land for agriculture can lead to exceedances of numeric criteria even in watersheds that currently are not managed (Ice and Binkley 2003).

Increases in stream nutrient concentration may follow timber harvest (Gravelle et al. 2009) and forest fertilization (Binkley et al. 1999), but these increases are typically small in magnitude and of limited duration. Harvest and fertilization occur infrequently in any particular stand (e.g., once or twice in a rotation period of 20–60 years), thus limiting their potential contributions to nutrient loads affecting lakes, estuaries, and other downstream waterbodies that may be sensitive to eutrophication. The dispersed nature of N and P inputs to streams in forested watersheds, including that via leaf fall directly into streams, suggests that attributing an excessive nutrient condition to any particular location or activity could be challenging, making remediation problematic.

Sediment

Nationwide, excessive sediment is a persistent problem that degrades drinking water quality and harms aquatic species (US EPA 2000), although there is variability across landscapes in the nature of this issue and implications for application of numeric criteria. In the southeastern United States, assessing the causes of excess sediment in streams is complicated by legacy effects associated with historic land use practices (Richter and Markewitz 2001), which make it difficult to identify sediment sources and their relative magnitudes. In the Murder Creek watershed in Georgia, for example, Jackson et al. (2005) documented that past farming practices led to deposition of massive quantities of topsoil in the stream channel and floodplain. This excess topsoil in stream channels is easily remobilized. In the western United States, there are many potential sources for fine sediment due to the mountainous terrain, but sediment yields associated with current harvest operations have been substantially reduced compared with historic practices (Gomi et al. 2005, Karwan et al. 2007). Across all regions of the United States, there have been large-scale efforts to develop and implement BMPs to reduce the potential for sediment to reach streams, including BMPs for road construction and maintenance, and substantial research efforts to evaluate BMP effectiveness (NCASI 2012).

Water quality standards for sediment often focus on suspended fine particles (generally a fraction of a millimeter in size) and their effects on optical properties of surface waters (i.e., turbidity). For example, in Washington, turbidity changes of 5 or 10 nephelometric units or 10–20% of background turbidity are allowed, depending on background turbidity levels and the stream habitat being protected [WAC-173-201a-200, Table 200 (1)(e), State of Washington Department of Ecology 2012). Similar turbidity standards have been developed by other states, but several problems have been identified in their application (Brown 1980, Markman 1990, NCASI 2002) due to natural variability in turbidity levels.

Recent studies by Ziegler (2002), Ankcorn (2003), and Riedel et al. (2003) document concerns about the potential limitations of turbidity as a water quality parameter to assess management impacts. As noted by these authors, detecting small changes in turbidity is prohibitively expensive, and there is a high potential for false-positive values in monitoring of upstream/downstream turbidity, given the tremendous natural variability exhibited by forest streams. Turbidity can increase after rain events due to resuspension of bottom sediments and bank erosion even if no sediment inputs via surface erosion result from forestry operations (Jackson et al. 2005).

In addition, sediment that penetrates stream buffers and reaches a waterbody is patchy in space and may occur only after large rain events in harvested areas with large contributing areas, the presence of old or reactivated agricultural gullies and steeper slopes (Rivenbark and Jackson 2004). Thus, attribution of violation is difficult. Biological effects, which include reducing light transparency for autotrophs (Lloyd et al. 1987) and influences on feeding ability of fish (Barrett et al. 1992) vary widely, depending on concentration and duration (Newcombe 2003, Diehl and Wolfe 2010). Collection of data on concentration and duration requires continuous measurement, which is costly. Finally, as a practical matter, it is difficult to monitor and regulate sediment discharges from roads that extend for many miles, often cross property boundaries, and are used by various kinds of vehicles for various purposes.

One conundrum associated with setting standards for sediment is that, as with nutrients, some sediment is environmentally beneficial. For example, high sediment-producing events, such as natural debris flows, can result in short-term impacts on water quality and habitat but can also have long-term benefits by creating off-channel habitat and restoring spawning gravels (Miller and Benda 2000). Controlled floods on the Colorado River are an example in which sediment-laden floods are actually a critical management tool for restoring aquatic habitat. In addition, the different particle sizes have different water quality impacts, cause biological impairment differently, and must be measured differently.

Dissolved Oxygen

The relationship between dissolved oxygen (DO) concentrations and aquatic productivity has long been recognized (Welch et al. 1988). Low DO can harm aquatic life and produce unacceptable water quality for human uses. States also recommend BMPs that limit the potential effects of forestry operations on DO. Relevant state BMPs include restrictions on placement of logging residues in stream channels and recommendations for its removal, guidelines that prevent application of fertilizers in or near streams, and retention of living trees and other vegetation in SMZs (Texas Forest Service 2000, North Carolina Division of Forestry 2006). As noted earlier, riparian buffers also reduce delivery of runoff-borne nutrients (which can increase biological oxygen demand) through immobilization, adsorption, and transformation processes.

Listings by the EPA of waterbodies as impaired due to low DO have rekindled interest in understanding factors and mechanisms contributing to violations of DO criteria in streams where influences of wastewater treatment plants are not apparent. For example, Ice and Sugden (2003) conducted a warm-season survey of 43 least-impaired and reference forest streams in Louisiana. They found that >80% of those stream reaches had locations with DO concentrations less than the criterion value of 5 mg/L and almost 60% of monitored sites had DO concentrations <3 mg/L. In an experiment of harvested and control sites in northern Louisiana, DO levels during low flow conditions were well below the EPA recommended standard at all sites and in all seasons (DaSilva et al. 2013). Of DO measurements, 83% were <1 mg/L at the control site in summer. DO was higher at the below-harvest downstream site (but still below the standard) because reduced evapotranspiration at the harvest site may have increased flow (DaSilva et al. 2013). Studies in the south have shown that DO levels <5 mg/L (a common numeric criterion) are frequent during the growing season even in areas with limited forest management or in watersheds with high percentages of forest (Carey et al. 2007, Todd et al. 2009, Florida Department of Environmental Protection 2011).

Multiple reasons have been hypothesized to explain the observed low DO levels, including low streamflow velocity and organic substrates (Ice and Sugden 2003, Todd et al. 2009), high growing season air and water temperatures (Joyce et al. 1985), and naturally high inputs of dissolved organic carbon (Meyer 1992). Although managers can avoid exacerbating low DO conditions by implementing BMPs (e.g., DaSilva et al. 2013), many streams appear naturally predisposed to DO concentrations below established numeric criteria. No forest management options are known that would increase naturally occurring low DO levels.

Temperature

Water temperature is an important aspect of habitat quality for fish and other aquatic organisms (Caissie 2006). High temperatures can also indirectly affect fish by reducing DO levels. Climate and weather obviously affect temperature regimes of surface waters including seasonal and diurnal patterns of variation. Additional factors affecting water temperature regimes include variation in volumes and sources of streamflow (e.g., snowmelt versus groundwater inflow), water exposure to solar radiation (e.g., effects of variation in stream width and shading by riparian vegetation), and human influences (e.g., stream impoundments, discharges from wastewater treatment plants, and forestry and agricultural activities in riparian areas). Timber harvest has long been linked to elevated temperatures in small forest streams (Brown and Krygier 1970), although when BMP buffers are used, impacts are less and may not persist downstream (Wilkerson et al. 2006, Gravelle and Link 2007).

Numeric temperature criteria for surface waters (e.g., temperature values not to be exceeded) may be designed to protect cold water fish species (e.g., McCullough 2011) and are sometimes viewed as rudimentary surrogates for fish habitat quality. However, aquatic habitat is diverse (e.g., riffle, glide, and pools) and fish can use different stream habitats in response to periodic temperature shifts. Stream pools, for example, can be important habitat for juvenile fish, especially where many stream reaches attain high summer temperatures and have dry season low flows (Nielsen et al. 1994). Not surprisingly, temperature criteria can be exceeded in many reference streams and other least impaired waters in Oregon, Washington, and Idaho (Ice et al. 2004b).

Research on temperature responses to forest management has resulted in BMPs to retain trees and other vegetation in riparian areas to maintain shade for stream surfaces (Ice et al. 1994). Vegetation in close proximity to a stream (e.g., within 10 m) provides most of the potential benefits of shade with respect to prevention of temperature increases after timber harvest (NCASI 2000). The ecological significance of these benefits, however, is uncertain because temperature increases in unshaded stream segments generally do not persist downstream of harvested areas (Zwieniecki and Newton 1999, Wilkerson et al. 2006, Gravelle and Link 2007), stream temperature varies naturally between parts of the stream, and natural regrowth of riparian vegetation begins to provide shade for streams within a few years after harvest (Andrus and Froehlich 1991).

Although stream temperature regimes can be easily measured, establishing relationships between stream temperature, BMP effectiveness, and habitat quality for aquatic species is inherently complex, especially for small headwater streams (Moore et al. 2005). Even if the effects of forest management on stream temperature regimes are not ecologically significant, they may still violate antidegradation elements of state water quality standards. Some states allow a *de minimis* increase in stream temperatures above "natural background conditions" after forest management activities, and this is probably achievable for large fish-bearing streams (Ice et al. 2004b). In non-fish-bearing streams, however, timber harvests may lead to exceedances in numeric criteria and standards may not be attainable even within unmanaged watersheds because of periodic natural disturbances (Ice and Schoenholtz 2003).

Conclusions

The development and widespread adoption of forestry BMPs to control silvicultural NPS activities is a well-documented example of how the CWA and state water quality agency programs have provided effective control of NPS pollution. All states with significant forestry activities have developed NPS control programs based on implementation of BMPs to minimize the potential for negative management impacts (Ice et al. 2010). Many states have conducted effectiveness monitoring and research to ensure that BMPs are achieving state water quality goals (NCASI 2012). BMPs are highly effective, widely implemented, and continually refined through the cooperative efforts of many stakeholders (Ice et al. 2010). When BMPs are widely implemented and effective, further progress toward meeting water quality criteria may be challenging (Clausen and Meals 1989).

Some participants in recent federal court cases have argued for replacing or supplementing NPS control programs based on BMPs with an approach that places greater emphasis on monitoring and regulating compliance with water quality standards. Some improvement in the specificity of governing regulations might help provide clarity on this topic. Within the current regulatory framework, there are also many challenges associated with applying this concept to forested streams. Numeric criteria are imperfect tools and may often be violated even in unmanaged forest watersheds (Argerich et al. 2013). When violations of criteria occur in managed forests, it may be difficult to distinguish the effects of forest management activities from the effects of natural variation, atmospheric deposition, and legacies of past land use. Determining causes of impairment can be problematic because water quality can be affected all along a stream length and not at any identifiable point, and routine monitoring in operational settings at many remote locations would be costly.

We suggest that effective application of water quality standards to forestry will require development and application of new methods for assessing relationships between numeric criteria and attainment of beneficial uses. As noted by a panel of the National Research Council (2001), developing more useful criteria will require a substantial research effort to better define variability in water quality parameters due to natural factors and human influences. Progress in this research will probably require improvements in the capability of current technology related to collection and statistical analysis of water quality data (National Research Council 2001).

We also suggest continued emphasis on evaluation of BMP effectiveness and, where appropriate, research to explore further refinements. For example, recent research by Wade et al. (2012) compared the efficacies of various approaches for bladed skid trail closure and identified several cost-effective options for improving water quality protection, e.g., water bars with piled hardwood or pine slash.

Future BMP research should measure not only improvements in water quality relative to past practices attributable to BMPs but also the effects of forestry operations with BMPs on beneficial uses of water bodies such as fish habitat support. Studies that link forestry BMPs to protection of beneficial uses of waterbodies would be costly and difficult to implement but may be necessary to maintain public support for monitoring BMP implementation as a practical and effective approach to ensuring stream protection in managed forests. The alternative may be endless debates about the ecological and legal significance of small changes in water quality parameters associated with forestry practices.

Endnotes

1. For more information on the Clean Water Act, please visit www.fws.gov/laws/lawsdigest/fwatrpo.html.
2. For more information, please visit water.epa.gov/polwaste/nps/whatis.cfm.
3. For an example in the Chesapeake Bay watershed, see www.epa.gov/reg3wapd/tmdl/ChesapeakeBay/tmdlexec.html.
4. See EPA Attains database at epa.gov/waters/ir/.
5. *Northwest Environmental Defense Center v. Brown*, 640 F.3d 1063 (9th Cir. 2011), cert. granted sub nom. *Decker v. Northwest Environmental Defense Center*, No. 11-338 (U.S.).
6. *National Cotton Council v. Environmental Protection Agency*, 553 F.3d 927 (6th Cir. 2009).
7. For more information, see water.epa.gov/scitech/swguidance/standards/criteria/nutrients/dataset_standards.cfm.

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RESPONSE

Forestry Best Management Practices: A Mitigated Water Pollution Success Story

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Forestry best management practices (BMPs) are a water pollution success story. The ideas behind BMP came from consideration of foresters' experiences combined with results and observations from scientific studies conducted mostly at the USDA Forest Service experimental watersheds in the 1950s, 1960s, and 1970s (e.g., Lieberman and Hoover 1948, Reinhart and Eschner 1962, Hewlett and Douglass 1968). The impetus for getting serious about BMP development and implementation came from the Clean Water Act (CWA), which set in motion a continuous feedback loop of BMP implementation, scientific assessments, and refinement. Taking my state of Georgia as an example, there were published forestry BMP manuals in 1981, 1999, and 2009, with intermediate updates in 1990, 1993, 2003, and 2005. Each manual and update featured refinements based on experience and assessment of the previous manuals. In the 1970s, the industry, land managers, and sci-

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