Lag Time in Water Quality Response to Best Management Practices: A Review

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Nonpoint source (NPS) watershed projects often fail to meet expectations for water quality improvement because of lag time, the time elapsed between adoption of management changes and the detection of measurable improvement in water quality in the target water body. Even when management changes are well-designed and fully implemented, water quality monitoring efforts may not show definitive results if the monitoring period, program design, and sampling frequency are not sufficient to address the lag between treatment and response. The main components of lag time include the time required for an installed practice to produce an effect, the time required for the effect to be delivered to the water resource, the time required for the water body to respond to the effect, and the effectiveness of the monitoring program to measure the response. The objectives of this review are to explore the characteristics of lag time components, to present examples of lag times reported from a variety of systems, and to recommend ways for managers to cope with the lag between treatment and response. Important processes influencing lag time include hydrology, vegetation growth, transport rate and path, hydraulic residence time, pollutant sorption properties, and ecosystem linkages. The magnitude of lag time is highly site and pollutant specific, but may range from months to years for relatively short-lived contaminants such as indicator bacteria, years to decades for excessive P levels in agricultural soils, and decades or more for sediment accumulated in river systems. Groundwater travel time is also an important contributor to lag time and may introduce a lag of decades between changes in agricultural practices and improvement in water quality. Approaches to deal with the inevitable lag between implementation of management practices and water quality response lie in appropriately characterizing the watershed, considering lag time in selection, siting, and monitoring of management measures, selection of appropriate indicators, and designing effective monitoring programs to detect water quality response.

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Over the past four decades, most watershed NPS projects have reported little or no improvement in water quality even after extensive implementation of conservation measures or best management practices (BMPs) in the watershed. Examples include the Lower Kissimmee River Basin in Florida, the Conestoga Headwaters in Pennsylvania, Oakwood Lakes-Poinsett in South Dakota, and Vermont's LaPlatte River Watershed and St. Albans Bay Watershed (Gunsalus et al., 1992; Koerkle, 1992; Goodman et al., 1992; Gale et al., 1993; Meals, 1993,1996; Jokela et al., 2004). Numerous factors contribute to the failure of such projects to achieve water quality objectives, including insufficient landowner participation, uncooperative weather, improper selection of BMPs or selection of BMPs for other than water quality purposes, mistakes in understanding of pollution sources, poor experimental design, and inadequate level or distribution of BMPs.

One important reason NPS watershed projects may fail to meet expectations for water quality improvement is *lag time*. Lag time is an inherent characteristic of the natural and altered systems under study that may be generally defined as the amount of time between an action and the response to that action. For this analysis, we define lag time as the time elapsed between installation or adoption of management measures at the level projected to reduce NPS pollution and the first measurable improvement in water quality in the target water body. Installation refers to the completion of the construction phase for structural practices. Adoption refers to the full use of an installed physical practice or management practice such as nutrient management. Even in cases where a program of management measures is well designed and fully implemented, water quality monitoring efforts—even those designed to be "long-term"—may not show definitive results if the monitoring period and sampling frequency are not sufficient to address the lag between treatment and response.

The objectives of this review are to explore the important components of lag time, to present examples of lag times reported from a variety of geographic regions and water resource settings, and to recommend ways for managers to cope with the lag between treatment and response in watershed projects.

Elements of Lag Time

Project management, system, and effects measurement components can be important determinants of lag between treatment

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Abbreviations: BMP, best management practice; NNPSMP, National Nonpoint Source Monitoring Program; NPS, nonpoint source.

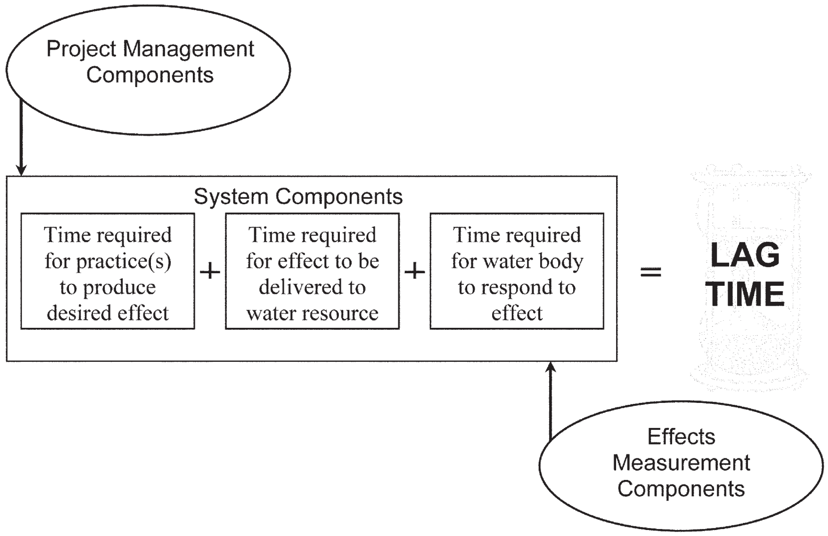

Fig. 1. Schematic showing the major elements of lag time in water quality response to best management practice (BMP) programs for nonpoint source (NPS) control. Planning process and measurement components are not part of a system lag in physical response, but often contribute to a perceived lag between action and result.

and response (Fig. 1). Any or all of these may come into play in a watershed project.

Project Management Components

The time required for planning and implementation of a NPS watershed project often causes the public to perceive a delay between the decision to act and results of that action. A project may be funded and announced today, but it will be some time—perhaps years—before that project will be fully planned and implementation begins. Even for point source control, it takes several years from the time a wastewater treatment plant upgrade is approved to when it is operational. The lag time from planning to implementation of NPS control practices can be even greater, considering the time required to identify NPS pollution sources and critical areas, design management measures, engage landowner participation, and integrate new practices into cropping and land management cycles. Clearly, delay due to planning and implementation is not an inherent system component of lag time as we have defined it; nevertheless, some stakeholders will experience it as part of the wait for results. Interested parties can be informed of planning and implementation delays through the education and outreach component of a logical and comprehensive watershed planning process (e.g., USEPA, 2005) that is absolutely critical for successful NPS projects.

Time Required for an Installed or Adopted Practice to Produce an Effect

Management practices (BMPs) are installed in watersheds to provide a wide range of effects to protect or restore the physical, chemical, and biological condition of waterbodies, including:

· Change hydrology

· Reduce dissolved pollutant concentration or load

· Reduce particulate/adsorbed pollutant concentration or load

· Improve vegetative habitat

· Improve physical habitat

The time required to produce these effects at the implementation site will vary depending on the degree of impairment and the appropriateness of the practices selected, as well as the nature of the effects themselves.

The time required for a BMP to be fully installed and become operational influences how quickly it can produce an effect. Concrete and steel treatment works may begin to function immediately after construction, with little time lag before pollutant discharge is reduced. Some NPS control measures may also become functional quickly. Installation of livestock exclusion fencing along several Vermont streams over a 3-mo period resulted in significant nutrient concentration and load reductions and reductions of fecal bacteria counts in the first post-treatment year (Meals, 2001). This response probably resulted from the combination of immediate prevention of new manure deposition in the stream and riparian zone and the availability of sufficient streamflow to flush residual manure through the system. However, other NPS management measures may take years to become fully effective. This is especially true of vegetative practices where plant communities need time to become established. For example, in a Pennsylvania study of a newly-constructed riparian forest buffer, the influence of tree growth on NO_3–N removal from groundwater did not become apparent until about 10 yr after tree planting (Newbold et al., 2008).

Lag time between BMP implementation and reduction of pollutant losses at the edge-of-field scale varies by the pollutant type and depends strongly on the behavior of the pollution source. Erosion controls such as cover crops, contour farming, and conservation tillage tend to have a fairly rapid effect on soil loss from a crop field as these practices quickly mitigate the forces contributing to detachment and transport of soil particles (Laflen et al., 1990).

However, the response time of runoff P to nutrient management is likely to be very different from that for soil loss and erosion control. Runoff losses of dissolved P are strongly controlled by soil P levels; very high soil P levels promote high levels of dissolved P in surface runoff (Pote et al., 1996; Sims et al., 2000). Where soil P levels are excessive, even if nutrient management reduces P inputs to levels below crop removal rates, it may take years to "mine" the P out of the soil to the point where dissolved P in runoff is effectively reduced (McCollum, 1991; Zhang et al., 2004; Sharpley et al., 2007).

Time Required for the Effect to be Delivered to the Water Resource

Practice effects initially occur at or near the practice location, yet managers and stakeholders usually want and expect the impact of these effects to appear promptly in the water resource of interest in the watershed. The time required to deliver an effect to a water resource depends on a number of factors, including:

- The route for delivering the effect
    - Directly in (e.g., streambed restoration) or immediately adjacent to (e.g., shade) the water resource
    - Overland flow (particulate pollutants)
    - Overland and subsurface flow (dissolved pollutants)
    - Infiltration groundwater and groundwater flow (e.g., nitrate)
- The path distance
- The path travel rate
    - Fast (e.g., ditches and artificial drainage outlets to surface waters)
    - Moderate (e.g., overland and subsurface flow in porous soils)
    - Slow (e.g., infiltration in absence of macropores and groundwater flow)
    - Very slow (e.g., transport in a regional aquifer)
    - Hydrologic patterns during the study period
    - Wet periods generally increase volume and rate of transport
    - Dry periods generally decrease volume and rate of transport

Once in a stream, dissolved pollutants like N and P can move rapidly downstream with flowing water to reach a receiving body relatively quickly. Even accounting for nutrient spiraling, the repeated uptake and release of nutrients by sediments, plants, or animals during downstream transport (Newbold et al., 1981), dissolved nutrients are not likely to be retained in a river or stream system for an extended period of time. Research in Vermont showed, for example, that despite active cycling of dissolved P between water, sediment, and plants in a river system, watershed P inputs to the river were unlikely to be held back from Lake Champlain by internal cycling for much more than 1 yr (Wang et al., 1999).

However, sediment and attached pollutants (e.g., P and some synthetic chemicals) can take years to move downstream as particles are repeatedly deposited, resuspended, and redeposited within the drainage network by episodic high flow events. This process can delay sediment and P transport (when P adsorbed to sediment particles constitutes a large fraction of the P load) from headwaters to outlet by years or even decades. Thus, substantial lag time could occur between reductions of sediment and P delivery into the headwaters and measurement of those reductions at the watershed outlet.

Pollutants delivered predominantly in groundwater such as nitrate N or some synthetic chemicals generally move at the rate of groundwater flow, typically much more slowly than the rate of surface water flow. About 40% of all N reaching the Chesapeake Bay, for example, travels through groundwater before reaching the Bay. Relatively slow groundwater transport introduces substantial lag time between reductions of N loading to groundwater and reductions in N loads to the Bay (Scientific and Technical Advisory Committee Chesapeake Research Consortium, 2005). The increased water storage created by summer fallow crop systems adopted in the 1940s led to the development of saline seeps in the Northern Great Plains that continued to expand into the 1970s (Miller et al., 1981). Groundwater nitrate concentrations in upland areas of Iowa were still influenced by the legacy of past agricultural management conducted more than 25 yr earlier (Tomer and Burkart, 2003).

Contaminant time of travel in groundwater is also influenced by the retardation factor, a term used to describe the delay in transport of a substance through the soil due to sorption, resulting in a net contaminant velocity less than the rate of groundwater flow (Rao et al., 1985). The retardation factor is a function of soil properties such as bulk density, organic carbon, and porosity and of contaminant characteristics such as sorption coefficient.

Time Required for the Water Body to Respond to the Effect

The speed with which a water body responds to the effect(s) produced by and delivered from management measures in the watershed introduces another increment of lag time. Hydraulic residence time (or the inverse, flushing rate), for example, is an important determinant of how quickly a water body may respond to changes in nutrient loading. Examples of residence times for selected North American lakes and estuaries are shown in Table 1. Simply on the basis of dilution, it will likely take considerably longer for water column nutrient concentrations to respond to a decrease in nutrient loading to Lake Superior (residence time 191 yr) than to Lake St. Clair (residence time 0.04 yr). Beyond dilution alone, residence time influences how a water body processes and exports nutrients. An analysis of annual nutrient budgets for North Atlantic estuaries demonstrated that the net fractional transport of N and P through estuaries to the continental shelf is inversely correlated with the residence time of water in the system (Nixon et al., 1996). Removal of N by denitrification and P loss by particulate settling were the primary processes responsible for removing nutrients, and the longer the water mass remains in the system, the greater the removal of N and P. Similarly, Dettmann (2001) used modeling and data from 11 North American and European estuaries to show that the fraction of N denitrified increases with increasing residence

Table 1. Hydraulic residence times for selected North American lakes and estuaries.

Waterbody	Location	Surface area	Residence time	Reference
		km^2	yr	
Lake St. Clair	MI/ONT	1114	0.04	Quinn, 1992
Narragansett Bay	RI	328	0.1	Dettmann, 2001
Delaware Estuary	DE	1989	0.3	Dettmann, 2001
Chesapeake Bay	VA/MD	11,542	0.6	Dettmann, 2001
Lake Erie	NY/PA/OH/ONT	25,700	3	Quinn, 1992
Lake Champlain	VT/NY/QUE	1127	3.3	LakeNet, 2009
Lake Mendota	WI	4	4.5	UW CFL, 2009
Lake Huron	MI/ONT	59,600	21	Quinn, 1992
Lake Michigan	MI/IN/IL/WI/ONT	57,800	100	Quinn, 1992
Lake Superior	MI/MN/ONT	82,100	191	Quinn, 1992
Lake Tahoe	CA/NV	501	650–700	UCD TERL, 2009

time, while the fraction of upland N input that is exported from the estuary decreases with increasing residence time.

Apparent lag time in water quality response may also depend on the indicator evaluated or the impairment involved, especially if the focus is on biological water quality. If *Escherichia coli* is the pollutant of concern, a relatively short lag time might be expected between reductions of bacteria inputs and reduction in bacteria levels in the receiving waters because the bacteria generally do not persist as long in the aquatic environment as do heavy metals or synthetic organic chemicals (Crane and Moore, 1986). While recent and ongoing research suggests that indicator bacteria may survive for months or more in aquatic sediments (Sherer et al., 1992) or soils (Byappanahalli et al., 2006), in the long run, without continual replenishment this stock would tend to be significantly diminished in a matter of months. The quantity of bacteria in the receiving water could therefore begin to reflect the incoming supply fairly quickly. Such response has been demonstrated in estuarine systems where bacterial contamination of shellfish beds has been reduced or eliminated through improved waste management on the land over a short period of time (Buzzards Bay National Estuary Program, 2008).

Improved sewage treatment in Washington, DC led to sharp reductions in point source P and N loading to the Potomac River estuary in the early 1970s (Jaworski, 1990). The tidal freshwater region of the estuary responded significantly over the next 5 yr with decreased algal biomass, higher water column dissolved oxygen levels, and increased water clarity.

While algae production might respond to changes in nutrient supply rapidly, ecosystem-level responses usually involve substantial lag time. Kemp et al. (2005) discussed the lag in response of the Chesapeake Bay estuary to changes in nutrient loading in terms of nonlinear feedback mechanisms. Enhanced nutrient release from anoxic sediments, for example, is an important positive feedback mechanism that reinforces the eutrophication process and could delay response to changes in nutrient loading. Conversely, water filtration from restored oyster reefs provides a negative-feedback control on eutrophication by reducing plankton biomass, increasing water clarity, and promoting growth of benthic plants. Enhanced sediment binding by benthic plants in turn helps maintain water clarity, allowing more light to support benthic photosynthesis. Such feedback control would reduce lag time in ecosystem response.

Macroinvertebrate or fish response to improved water quality and habitat conditions in stream systems requires time for the organisms to migrate into the system and occupy newly improved habitat. Significant lag times have been observed in the response of benthic invertebrates and fish to management measures implemented on land, including in the Middle Fork Holston River project (Virginia), where Index of Biotic Integrity (IBI, a measure of the stream fish community) scores and *Ephemeroptera-Plecoptera-Trichoptera* (EPT, a measure of the benthic macroinvertebrate community) scores did not improve, even though the project accomplished substantial reduction in the sediment, N, and P loadings (Virginia Department of Conservation and Recreation, 1996). The lack of increase in the biological indicator scores indicates a system lag time between the actual BMP implementation and positive changes in the biological community structure. This lag could depend in part on the amount of ecological connectivity with neighboring healthier aquatic systems that could provide sources of appropriate organisms to repopulate the restored habitats.

Exceptions to such lag in response of stream biota can occur where in-stream restoration is the BMP applied. Restoration of stream habitat, particularly the temperature regime, in Oregon's Upper Grand Ronde River system led to significant increases in rainbow trout numbers within 2 yr (Whitney and Hafele, 2006). Naturalization of stream channel morphology and enhancement of habitat in the Waukegan River (Illinois) involved vegetative and structural stabilization and habitat structures including a series of pool-and-riffle complexes using stone weirs. Significant improvement in habitat, macroinvertebrate communities, and in the number and abundance of fish species were documented quickly in the study reach, although full biological restoration in the Waukegan River will require more comprehensive efforts to address other stressors such as water quality impairment and hydrologic discharge extremes (Roseboom et al., 1996; White et al., 2003).

In several Vermont streams, the benthic invertebrate community improved within 3 yr in response to reductions of sediment, nutrient, and organic matter inputs from the land (Meals, 2001). However, despite observed improvement in stream physical habitat and water temperature, no improvements in the fish community were documented. The project attributed this at least partially to a lag time in community response exceeding the monitoring period.

Even when reductions of tributary pollutant loads are observed in a short time, the variable response times of receiving water bodies may introduce a significant lag time between load reductions and restoration of impaired uses. In some cases, this lag time may be relatively short. For example, researchers anticipate that the Chesapeake Bay could respond fairly rapidly to reductions in nutrient loading, as incoming nutrients are quickly buried by sediment or exported to the atmosphere or the ocean. Even beds of submerged aquatic vegetation, critical to the Bay's aquatic ecosystem, can return within a few years after improvements in water clarity (Scientific and Technical Advisory Committee Chesapeake Research Consortium, 2005). Bacteriological water quality in shellfish beds in Totten and Eld Inlets (WA) estuaries improved rapidly in response to improved animal waste management in the drainage area, but unfortunately also deteriorated equally rapidly when animal waste management practices on the land were abandoned (Spooner et al., 2008).

In contrast, lake response to changes in incoming P load is often delayed by recycling of P stored in aquatic sediments. When P loads to Shagawa Lake (MN) were reduced by 80% through tertiary wastewater treatment, residence time models predicted new equilibrium P concentrations within 1.5 yr, but high in-lake P levels continued to be maintained by recycling of P from lake sediments (Larsen et al., 1979). Even more than 20 yr after the reduction of the external loading, sediment feedback of P continued to influence the trophic state of the lake (Seo and Canale, 1999).

Similarly, St. Albans Bay (VT) in Lake Champlain failed to respond rapidly to reductions in P load from its watershed. From 1980 through 1991, a combination of wastewater treatment upgrades and intensive implementation of dairy waste management BMPs through the Rural Clean Water Program (RCWP) brought about a reduction of P loads to this eutrophic bay. However, water quality in the bay did not improve significantly; this pattern was attributed to internal loading from sediments highly enriched in P from decades of point and NPS inputs (Meals, 1992). Although researchers at that time believed that the sediment P would begin to decline over time as the internal supply was depleted, subsequent monitoring has shown that P levels have not declined over the years as expected (LCBP, 2008). Recent research has confirmed that a substantial reservoir of P continues to exist in the sediments that can be transferred into the water under certain chemical conditions and nourish algae blooms for many years to come (Druschel et al., 2005). In effect, this internal loading has become a significant source of P, one that cannot be addressed by management measures on the land.

Effects Measurement Components of Lag Time

Watershed project managers are routinely pressed for results by a wide range of stakeholders. The fundamental temporal components of lag time control how long it will take for a response to occur, but the effectiveness of measuring the response may cause a further delay in recognizing it. It is possible for a response to occur unnoticed unless a suitable monitoring program is in place to detect it.

The magnitude of the potential effects produced by a watershed NPS management program depends on the effectiveness of each unit of installed or adopted practices, the number of practice units installed or adopted, the effectiveness with which the practices are targeted to the correct pollutants and sources, and numerous other factors. While not all responses can be measured, the design of the monitoring program is a major determinant of our ability to discern a response against the background of the variability of natural systems.

In the context of lag time, sampling frequency with respect to background variability is a key determinant of how long it will take to document change. In a given system, taking n samples per year provides a certain statistical power to detect a trend. If the number of samples per year is reduced, statistical power is reduced, and it may take longer to document a significant trend or to state with confidence that a concentration has dropped below a water quality standard. Simply stated, taking fewer samples a year is likely to introduce an additional "statistical" lag time before a change can be effectively documented.

In an analysis of a monitoring program for the San Diego County (California) MS4 permit, Weston Solutions, Inc. (2005) assessed the ability of a stormwater monitoring program to continue to detect trends in receiving waters under different sampling frequencies. This analysis used existing data and projected trends into the future using the slope of the current data set to predict when concentration of a constituent (based on a trend line from linear regression against time) would drop below the water quality objective. An example of this analysis for total Cu concentration is shown in Fig. 2. The current program design of three events each year is predicted to show Cu concentration dropping below the water quality objective in 2009 (Fig. 2A); an alternative program monitoring two storm events every other year will not confirm Cu concentrations below the water quality objective until 2018 (Fig. 2.B.), adding 9 yr of statistical lag time before a response to stormwater management can be documented with statistical confidence.

The Magnitude of Lag Time

The magnitude of lag time is difficult to predict in specific cases and generalizations are difficult to make. A few examples, described below and summarized in Table 2, can illustrate some possible time frames for several categories of lag times.

Sediment storage in stream systems has been widely observed to introduce lag time into sediment input/output analyses. Marutani et al. (1999) reported that two New Zealand watersheds responded to extreme storm events by rapidly aggrading, then gradually flushing out the temporarily stored sediment. The authors estimated that 1.2 to 2.75 × 10^6 m^3 of sediment were delivered to the stream systems in a 1988 flood, while subsequent scouring rates were 0.8 to 1.25 × 10^4 m^3 yr^{-1}, suggesting the time needed to remove all the stored sediment—absent any additional sediment input—could be approximately 8 to 25 yr. Clark and Wilcock (2000) concluded that coarse sediment delivered to a river system in a Puerto Rican watershed from extensive land clearing and agriculture from 1830 to 1950 continued to influence the river system in 2000.

Newson (2007) conducted intensive surveys and simulation modeling to quantify the response of the sediment budget of Paradise Creek, ID to implementation of agricultural conservation practices in the watershed uplands. The study concluded that from 1978 to 1980, almost 2000 t of sediment were deposited in the lower stream region, nearly all of it originating from upstream reaches and hillslopes. Over the next 16 yr (1982–1998) this sediment was gradually recovered and transported out of storage and exported from the watershed. Thus, sediment entering this part of the stream channel in 1978–1980 had an estimated 19-yr time lag before being completely moved beyond the watershed outlet.

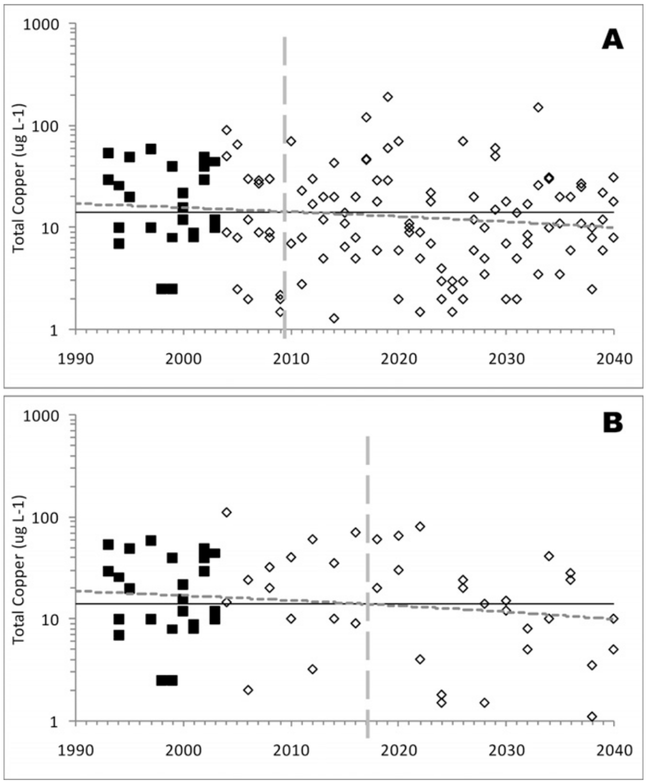

Fig. 2. Plots of a trend assessment for total Cu at Tecolote Creek (CA). Solid squares represent actual data collected 1992 to 2003; open diamonds represent simulated data based on extrapolating the existing trend to 2040. Panel A shows simulated data collected with the existing program of three storm events per year, 2004 to 2040; Panel B shows a scenario of a reduced sampling frequency of two storm events in alternate years, 2004 to 2040. The horizontal line represents the Water Quality Objective (WQO) for total Cu of 13 μg L^{-1}. In each panel, the vertical dashed line represents the year in which the dashed trend line (linear regression through all the data) is predicted to cross the WQO at P $\leq$ 0.05. (adapted from Weston Solutions, Inc. 2005, used by permission).

A long-term Pennsylvania project evaluated the development and performance of a newly established riparian forest buffer (Newbold et al., 2008; Spooner et al., 2008). Substantial reforestation of the riparian area took 8 to 12 yr (Fig. 3A), considerably longer than anticipated due to drought and deer (*Odocoileus virginianus*) damage. Preliminary analysis of groundwater nitrate data indicates that, except for initial reductions due to removing agriculture from the buffer area, significant nitrate removal from groundwater flowing toward the stream did not occur until trees were fully established and dominated the site 10 yr after planting. (Fig. 3B). The 1992 to 1997 increase in nitrate in all sectors shown in Fig. 3B likely reflects higher fertilizer application during this period than during the period before the study (Newbold et al., 2008). The results of the project so far suggest that water quality improvement from riparian reforestation may take on the order of a decade or more to be measurable in the stream.

Effective reduction of elevated soil test P levels and runoff P concentrations from such soils appears to involve lag times on the order of a decade or more. In plot studies of P fertilization and soil P depletion under continuous corn (*Zea mays* L.), Zhang et al. (2004) reported a Mehlich-3 soil test P decline of 3.96 mg P kg^{-1} yr^{-1} following cessation of P fertilization. The authors estimated that it would require about 28 yr to deplete the 110 mg P kg^{-1} of soil P built up during 6 yr of high-rate P fertilizer addition to base-line levels.

McCollum (1991) described the decay of available P in soil following cessation of P fertilization as having the form of a first-order chemical reaction with a rate constant proportional to the initial soil P content. A Portsmouth soil starting at 50 to 60 g P m^{-3} (Mehlich-1) dropped to 22 g P m^{-3}, the approximate yield-limiting level for corn, in 8 to 10 yr without further P additions. The same soil with 100 to 120 g P m^{-3} initial soil test P declined to 22 g P m^{-3} in about 14 yr.

Sharpley et al. (2007) described the response of soil P and runoff P to poultry litter nutrient management documented in two studies. In Oklahoma bermudagrass (*Cynodon* spp.) plots, 3 yr of litter applications raised soil test P and P concentrations in surface runoff and subsurface flow dramatically. In 6 yr following the end of litter applications, soil test P (Mehlich-3) declined from over 250 mg P kg^{-1} to about 125 mg P kg^{-1} and dissolved P in surface runoff had declined from over 4 mg P L^{-1} to 0.22 mg P L^{-1}. However, the soil test P level was still considerably higher than crop requirements (20–60 mg P kg^{-1}) and runoff concentrations were higher than surface water thresholds associated with accelerated eutrophication (0.02–0.05 mg P L^{-1}). In a study of poultry litter application to a corn–soybean [*Glycine max* (L.) Merr.] rotation in Maryland's Eastern Shore, implementation of P-based nutrient management reduced both soil test P (Mehlich-3) and runoff P, but it took 3 yr for the effect to become evident and even after 5 yr of nutrient management, both mean annual total P runoff concentrations (1.07–1.85 mg P L^{-1}) and soil test P values (320–480 mg P kg^{-1}) were still considerably above environmental thresholds of 0.05 mg P L^{-1} for flowing waters and 75 mg P kg^{-1} for soils.

On highly enriched soils (394 kg P ha^{-1} Mehlich-3, ~25% P saturation) supporting corn, soybean, and wheat (*Triticum aestivum* L.) in Quebec, Canada, Giroux and Royer (2007) reported that 8 yr after P fertilizer rates were reduced, soil test P values were 270, 281, and 294 kg P ha^{-1} for 0, 30, and 60 kg P_2O_5 ha^{-1} application rates. Because of the balance between P added and P removed in harvested crops, the regulatory target P saturation value of 13.1% was achieved after 10 yr for the 0 kg P_2O_5 ha^{-1} rate, 12 yr for the 30 kg P_2O_5 ha^{-1} rate, and 14 yr for the 60 kg P_2O_5 ha^{-1} rate.

The rate of groundwater movement and pollutant transport can be a major contributor to lag time in water quality response to management practices. On a small scale, hydrologic modifications to the recharge of the groundwater system may yield rapid response. In dry-land farming regions, for example, saline seeps can develop when excess water percolates below the root zone, dissolves salts in the subsoils, accumulates on an impermeable layer, and deposits the salts in a surface discharge area. Summer fallow cropping rotations, adopted extensively in the 1940s, promoted the development of saline seeps by decreasing water uptake from the soil and increasing recharge in vulnerable areas. Because of low rainfall, saline seeps took decades to develop; in Montana, for example, areas affected by seeps grew from 32,000 ha in 1971 to 81,000 ha in 1978 (Miller et al., 1981). Methods to control saline seeps include cropping systems with deep-rooted perennial vegetation, flexible

Table 2. Examples of lag times reported in response to environmental impact or treatment.

Parameter(s)	Scale	Impact/Treatment	Response lag	Reference
Sediment	Large watershed	Extreme storm events	8–25 yr	Marutani et al., 1999
Sediment	River basin	Land clearing/agriculture	> 50 yr	Clark and Wilcock, 2000
Sediment	Large watershed	Cropland erosion control	19 yr	Newson, 2007
Chloride	Large aquifer	Road salt	> 50 yr	Bester et al., 2006
Salinity	Field	Hydrologic control/alfalfa cropping	5 yr	Halvorson and Reule, 1980 Miller et al., 1981
NO_3–N	Small watersheds	N fertilizer rates	> 30 yr	Tomer and Burkart, 2003
NO_3–N	River basin	N fertilizer rates	> 50 yr	Bratton et al., 2004
NO_3–N	Large watersheds	Nutrient management	≥ 5 yr	STAC, 2005
NO_3–N	Small watershed	Nutrient management	15–39 yr	Galeone, 2005
NO_3–N	Small watershed	Prairie restoration	10 yr	Schilling and Spooner, 2006
NO_3–N	Small watershed	Riparian forest buffer	10 yr	Newbold et al., 2008
NO_3–N	Small watersheds	N fertilizer rates	4–10 yr	Owens et al., 2008
Soil test P	Field	P fertilizer rates	8–14 yr	McCollum, 1991
Soil test P	Plot	P fertilizer rates	28 yr	Zhang et al., 2004
Soil test P	Field	P fertilizer rates	10–14 yr	Giroux and Royer, 2007
Soil and runoff P	Plot/field	Poultry litter management	> 5 yr	Sharpley et al., 2007
P	Lake	WWTP upgrade	> 20 yr	Larsen et al., 1979
P	Lake	WWTP upgrade/agricultural BMPs	> 20 yr	LCBP, 2008
P, N, *E. coli*	Small watersheds	Livestock exclusion	≤ 1 yr	Meals, 2001
Fecal bacteria	Estuary	Waste management	< 1 yr	BBNEP, 2008
Fecal bacteria	Estuary	Waste management	1 yr	Spooner et al., 2008
Algal biomass, dissolved oxygen, water clarity	Estuary	Wastewater treatment	< 5 yr	Jaworski, 1995
Macroinvertebrates	Small watersheds	Livestock exclusion	3 yr	Meals, 2001
Macroinvertebrates	Small watersheds	Mine waste treatment	10 yr	Chadwick et al., 1986
Fish	first order stream	Habitat restoration	2 yr	Whitney and Hafele, 2006
Fish	Large watershed	Conservation Reserve Program (before/after)	25 yr	Marshall et al., 2008
Fish	Small watershed	Acid mine drainage treatment	3–9 yr	Cravotta, 2009

crop rotations, and surface drainage of upland recharge areas to reduce the amount of water percolating through the root zone. In Montana, hydrologic controls, including planting of alfalfa (*Medicago sativa* L.) dropped the water table up to 200 cm and effectively dried out the subsoil, reducing hydraulic pressures from the recharge to the discharge areas, eliminating saline discharge in < 5 yr (Halvorson and Reule, 1980; Miller et al., 1981).

While groundwater levels may respond relatively quickly to changes in vegetation, transport of solutes in groundwater can take considerably longer, especially as the size of the system increases. For example, Delaware's Inland Bays, comprising a 77-km^2 estuary on the state's southern Atlantic coast, suffer from excessive nutrient and sediment loading, resulting in degraded communities of benthic organisms, submerged vegetation, and fish. Nitrate delivered to the Bays in groundwater discharge from agricultural fields and poultry operations and from septic-system effluent in the 830-km^2 watershed is one of the most severe stressors of the Inland Bays. Studies and efforts to reduce nitrate loading have been underway for two decades, from state, university, and USGS groundwater studies in the 1970s to a USDA Hydrologic Unit Area (HUA) project in the 1990s to a total maximum daily load (TMDL) in 2004. Unfortunately, estuary response to restoration efforts is constrained by the estimated 50 yr required for groundwater to travel from agricultural land in the watershed to the Bays (Bratton et al., 2004; Krantz et al., 2004).

In the Pequea and Mill Creek Watershed (Pennsylvania) Clean Water Act Section 319 National Nonpoint Source Monitoring Program (NNPSMP) Project (1994–2003), changes in fertilizer applications to cropland in a 3.6 km^2 watershed did not result in reductions in stream N concentrations due to lag time between applications and nutrients reaching stream channel. Groundwater age dating conducted during the study indicated that N applied to land reached springs in 2 to 3 yr, but groundwater flow to the stream channel took 15 to 39 yr (Galeone, 2005).

Tomer and Burkart (2003) presented extensive evidence from two ~30 ha Iowa agricultural watersheds that at least several decades were required for subsurface water to travel from the watershed divide to the stream and concluded that, in many watersheds, changes in agricultural practices may take several decades to fully effect changes in groundwater quality. In their study watersheds, for example, groundwater concentrations of NO_3–N in 2003 were still influenced by heavy N fertilizer applications that occurred in the 1970s.

Researchers in the Walnut Creek Restoration (Iowa) NNPSMP Project (1995–2005) conducted a groundwater travel time analysis using a geographic information system (GIS) and readily available soil and topographic data to evaluate the time needed to observe changes in stream nitrate N concentrations resulting from conversion of row crop land to native prairie (Schilling and Wolter, 2007). Mean groundwater travel time in the 7.8 km^2 watershed was estimated to be 10.1 yr, with a range from 2 d to 308 yr. Researchers estimated that 10 to 22% of restored prairie areas contributed groundwater to streams in the Walnut Creek watershed within the 10-yr

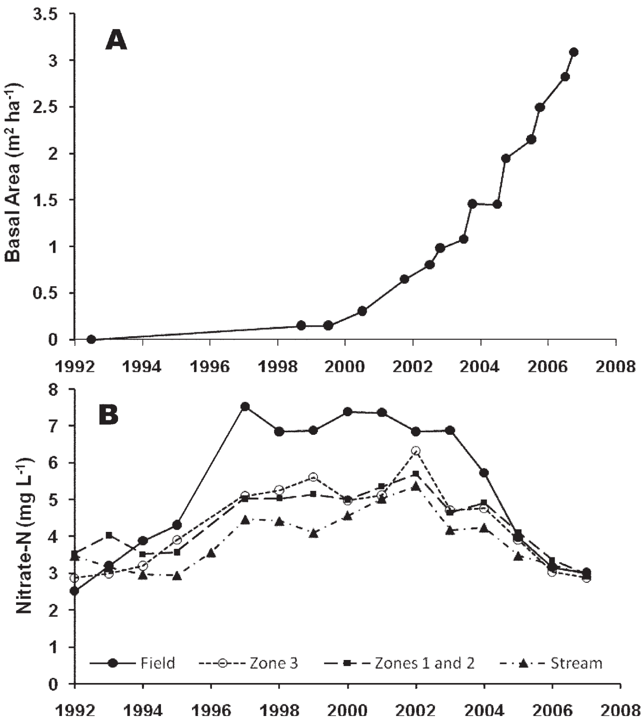

Fig. 3. A. Plot of changes in basal area of trees in Zone 2 of a planted riparian forest buffer system in Pennsylvania, 1992 to 2007. B. Plot of mean annual nitrate concentrations in groundwater and stream water in the riparian forest buffer system, 1992 to 2007. "Field" refers to the corn field draining to the buffer, "Zones 1 and 2" are the forested areas closest to the stream, and "Zone 3" refers to the herbaceous or grass filter strip between the forested zone and the upland field. Standard deviations for individual points averaged 1.35 (range: 0.41–2.0) mg L^{-1} for groundwater and 0.61 (range: 0.25 to 1.2) mg L^{-1} for stream water. Sample sizes ranged from 15 to 28 per year (adapted from Newbold et al. 2008, used by permission).

project period. Despite this relatively small contribution, the project was able to document significant reductions in stream nitrate concentrations in response to prairie restoration (Schilling and Spooner, 2006) and researchers anticipate additional nitrate reduction as reduced-nitrate groundwater from additional watershed area reaches the stream network.

Owens et al. (2008) reported that groundwater quality response to changes in N fertilization rate took several years to observe even in very small (< 2 ha) watersheds. When N application rate was increased from 56 to 168 kg ha^{-1} yr^{-1}, NO_3–N concentrations in groundwater and streamflow took 4 yr to respond, and then continued to increase for 10 yr. After N additions were discontinued, NO_3–N in groundwater discharge did not return to pre-treatment levels for 6 yr. Even where clay layers forced groundwater discharge pathways to be very shallow, suggesting a potentially rapid response to changes in N inputs, the lag in groundwater NO_3–N concentration response to management changes was still 3 yr or more.

Research in the Chesapeake Bay Watershed has emphasized the likelihood of a substantial lag time between implementation of BMPs and reductions in N loading to the Bay (Phillips and Lindsey, 2003; Scientific and Technical Advisory Committee Chesapeake Research Consortium, 2005). Groundwater supplies a significant amount of water and N to streams in the watershed and about half of the N load in streams in the Bay watershed is thought to be transported through groundwater. The age of groundwater in shallow aquifers in the Chesapeake Bay watershed ranges from <1 to more than 50 yr. The median age of all samples was 10 yr, with 25% of the samples having an age of 7 yr or less and 75% of the samples having an age of up to 13 yr. Based on this age as representative of time of travel, scientists estimated that in a scenario of complete elimination of N applications in the watershed, a 50% reduction in base flow nitrate concentrations would take about 5 yr, with equilibrium reached in about 2040.

Bester et al. (2006) studied the impact of road salt on a municipal wellfield outside Toronto, ON, using monitoring data and numerical simulation. Their results suggested that the aquifer system contains a large and heterogeneously distributed mass of chloride and that some of the wells may not yet have reached their maximum chloride concentrations even after 57 yr (1945–2002) of road salt application. Conversely, the simulations indicated that although the system responds rapidly to reductions in salt loading, the residual chloride mass may take decades to flush out, even if road salting were discontinued. Under conditions of continuous salt input, attainment of equilibrium concentrations in the system may require on the order of 100 yr.

In sum, at best only broad ranges of lag times can be generalized. In the Chesapeake Bay watershed, where lag time issues have been examined closely, Phillips and Lindsey (2003) proposed some general guidelines for considering lag times in the Bay restoration program (Table 3).

Discussion

Several important factors affect lag time. **Scale** is clearly an important influence; intuitively, response to BMPs should be more rapid at small scales than in larger settings. The relationship between scale and response, however, does not appear to be consistent (see Table 2). Note that lag time for soil test P response to change in P fertilizer rate was 28 yr in a plot study (Zhang et al., 2004); while lag time for P, N, and *E. coli* response to livestock exclusion in small watersheds was 1 yr or less (Meals, 2001). **Physical processes**, such as sediment transport in streams, affect the rate at which response to a perturbation or a management improvement is delivered to a water body. **Chemical processes**, such as sorption kinetics in soils or aquatic sediments, have often delayed expected responses to changes in pollutant loading. **Groundwater movement** frequently controls the rate at which changes in pollutant loads are delivered to receiving surface waters. The **type of management**, including selection of appropriate BMPs, application of BMPs to critical source areas, and achievement of a sufficiently high level of treatment to effect change, will affect the nature and speed of response in water quality. Selection of an **indicator** and the design of the **monitoring system** help determine how and when a water quality response to changes in land management can be detected.

In most situations, some lag time between implementation of BMPs and water quality response is inevitable. Although the exact duration of the lag can rarely be predicted, in many cases the lag time will exceed the length of typical monitoring periods,

Table 3. Guidelines for considering lag times in the Chesapeake Bay restoration program (Phillips and Lindsey, 2003).

Nutrient source	Management practice implementation time	Watershed residence time	Implications for load reductions to Chesapeake Bay
Point sources	Several years	Hours to weeks	Would provide most rapid improvement of water quality due to immediate reduction of source.
Dissolved nutrients from nonpoint sources	Several years	Hours to months if associated with runoff/soil water Years to decades if associated with groundwater (median time 10 yr)	Once fully treatments fully implemented, there would be a fairly rapid reduction of loads associated with runoff and soil water. Nitrogen loads associated with groundwater would have a median time of 10 yr for water quality improvements to be evident.
Sediment-associated nutrients from nonpoint sources	Several years	Decades or longer, depending on location in watershed	Load reductions would be greatly influenced by streamflow variability. Storm events would deliver sediment and associated nutrients contained on land and in stream corridors to the Bay may not show reductions for decades due to long residence times.

making it problematic to document a water quality response. Several possible approaches are proposed to deal with this challenge.

Recognize Lag time and Adjust Expectations

It usually takes time for a water body to become impaired and it will take time to accomplish the clean-up. However, once a water quality problem is recognized and action is taken, the public and political system usually expect quick results. Failure to meet such expectations may cause frustration, pessimism, and a reluctance to pursue further action. It is up to scientists, investigators, and project managers to do a better job explaining to all stakeholders in realistic terms that current water quality impairments usually result from historically poor land management and that immediate solutions should not be expected. For example, a 2005 report (Scientific and Technical Advisory Committee Chesapeake Research Consortium, 2005) advised the Chesapeake Bay Program to better communicate the implications of the lag time between management actions, watershed properties, and cycles in weather conditions on the restoration of the Chesapeake Bay, noting that effective communication of this point will be very important for continued support of efforts to implement BMPs even though the results of these actions often will not be immediately observed.

Characterize the Watershed

Before designing a NPS management program and an associated monitoring program, investigate important watershed characteristics likely to influence lag time. Determining the time of travel for groundwater movement is an obvious example. Watershed characterization is an important step in the project planning process (USEPA, 2005) and such characterization should especially address important aspects of the hydrologic and geologic setting, as well as documentation of NPS pollution sources and the nature of the water quality impairment, all of which can influence observed lag time in system response.

Consider Lag Time Issues in Selection, Siting, and Monitoring of Best Management Practices

First and foremost, proper BMP selection must be based on solving the problem and ensuring that landowners have the capability and willingness to implement and maintain the BMPs. Lag time can, however, be an important factor in the final design of BMP systems, ensuring, for example, that when down-gradient BMPs are installed, they are ready to handle the anticipated runoff or pollutant load from up-gradient sources. In addition to that, when projects include targeted BMP monitoring to document interim water quality improvements, recognition of lag time may require an adjustment of the approach to targeting the management program. When designing a program for projects that include BMP-specific monitoring, potential BMPs should be evaluated to determine which practices might provide the most rapid improvement in water quality, given watershed characteristics. For example, practices affecting direct delivery of nutrients into surface runoff and streamflow, such as barnyard runoff management, may yield more rapid reductions in nutrient loading to the receiving water than practices that reduce nutrient leaching to groundwater, when groundwater time of travel is measured in years. Fencing livestock out of streams may give immediate water quality improvement, compared to waiting for riparian forest buffers to grow. Such considerations, combined with application of other criteria such as cost effectiveness, can help determine priorities for BMP programs in a watershed project.

Lag time should also be considered in locating management practices within a watershed. Managers should consider the need to demonstrate results to the public, which may be easier at small scales, along with the need to achieve water quality targets and consequent wider benefits at the large watershed scale. Where sediment and sediment-bound pollutants from cropland erosion are primary concerns, for example, implementing practices that target the largest sediment sources closest to the receiving water may provide a more rapid water quality benefit than erosion controls in the upper reaches of the watershed. Where groundwater transport is a key determinant of response, application of a groundwater travel time model such as that used in Walnut Creek, Iowa (Schilling and Wolter, 2007) **before** application of management changes could at least help managers understand when to anticipate water quality response and communicate this issue to the public, or at best support targeting of an initial round of application of management measures to land areas where the effects are expected to be transmitted to receiving waters quickly.

It is important to point out that factoring lag time into BMP selection and targeting is not to say that long-term management improvements like riparian forest buffer restoration should be discounted or that upland sediment sources should be ignored. Rather, it is suggested that planners and managers may want to consider implementing BMPs and treating sources likely to exhibit short lag times first to increase the probability of demonstrating some water quality improvement as quickly as possible. "Quick-fix" practices with minimum lag time should not automatically replace practices implemented in locations that can ultimately yield permanent reductions in pollutant loads.

Monitor Small Watersheds Close to Sources

In cases where documentation of the effects of a management program on water quality is a critical goal, lag time can sometimes be minimized by focusing monitoring on small watersheds, close to pollution sources. Lag times introduced by transport phenomena (e.g., groundwater travel, sediment flux through stream networks) will likely be shorter in small watersheds than in larger basins. In the extreme, this principle implies monitoring at the edge of field or above/below a limited treated area, but small watersheds (e.g., < 1500 ha) can also yield good results. In the NNPSMP, projects monitoring BMP programs in small watersheds (e.g., the Morro Bay Watershed Project in California, the Jordan Cove Project in Connecticut, the Pequea/Mill Creek Watershed Project in Pennsylvania, and the Lake Champlain Basin Watersheds Project in Vermont) were more successful in documenting improvements in water quality in response to change than were projects that took place in large watersheds (e.g., the Lightwood Knot Creek Project in Alabama and the Sny Magill Watershed Project in Iowa) in the 7 to 10 yr time frame of the NNPSMP (Spooner et al., 2008).

Monitoring programs can be designed to get a better handle on lag time issues. Monitoring indicators at all points along the pathway from source to response or conducting periodic synoptic surveys over the course of a project will identify changes as they occur and document progress toward the end response. Special studies of sediment transport, soil P levels, groundwater dynamics, or receiving water behavior can shed light on phenomena that affect lag time in water quality response. For example, the Long Creek Watershed (NC) NNPSMP Project (1993–2002) conducted special studies of the effects of a wetland on PAH concentrations in an urban stream, the use of microbial indicators to assess land use impacts, and interactions between P and stream sediments to better explain the temporal and spatial water quality response to a BMP program (Line and Jennings, 2002). Supplementing a stream monitoring program with special studies can help project managers understand watershed processes, predict potential lag times, and help explain delays in water quality improvement to stakeholders. In the Walnut Creek (Iowa) watershed, no changes in stream suspended sediment loads were documented, despite extensive conversion of row crop land to prairie and reductions in field erosion predicted by RUSLE; a 22-mile stream survey revealed that streambank erosion contributed more than 50% of Walnut Creek sediment export (Spooner et al., 2008).

Select Indicators Carefully

Some water quality variables can be expected to change more quickly than others in response to management changes. For example, Tomer and Burkart (2003) recommended that shallow monitoring of unsaturated-zone waters, rather than in-stream monitoring, may be the most reliable means to document the effects of changes in crop rotations and fertilization on water quality. As documented in the Jordan Cove (CT) NNPSMP Project (1996–2005), peak storm flows from a developing watershed can be reduced quickly through application of stormwater infiltration practices (Clausen, 2004). Reductions in nutrient loads in surface waters might be expected to occur promptly in response to a ban on winter application of animal waste in northern states. The NNPSMP projects in California, North Carolina, Pennsylvania, and Vermont (Spooner et al., 2008) demonstrated rapid reductions in nutrients and bacteria by reducing direct deposition of livestock waste in surface waters through fencing livestock out of streams. However, improvements in stream biota appear to come much more slowly, beyond the time frame of many monitoring efforts. Where restoration of biological integrity is a goal, this may argue for a more sustained monitoring effort to document a biological response to land treatment. Failing that, however, selection of indicators that have relatively short lag times where possible will make it easier (and quicker) to demonstrate success.

Incorporate Lag Time into Simulation Modeling

In long-term restoration efforts in complex systems, simulation models are often used for program planning, forecasting, and evaluation. However, most current models do not handle the lag time issue well; significant improvements are needed before models can represent actual landscape processes to provide more realistic predictions of water quality changes.

Design Monitoring Programs to Detect Change Effectively

Monitor at locations and at a frequency sufficient to detect change with reasonable sensitivity. Assess background variability before the project begins and conduct a minimum detectable change analysis (Spooner et al., 1987; Richards and Grabow, 2003) to determine a sampling frequency sufficient to document the anticipated magnitude of change with statistical confidence. Although lag time will still be a factor in actual system response, a paired-watershed design (Clausen and Spooner, 1993; King et al., 2008), where data from an untreated watershed are used to control for weather and other sources of variability, is one of the most effective ways to document water quality changes in response to improvements in land management. If a monitoring program is intended to detect trends, evaluate statistical power to determine the best sampling frequency for the project.

Target monitoring to the effects expected from the BMPs implemented, in the sequence that those effects are anticipated. For example, when the ultimate goal is habitat/biota restoration in an urban stream, if BMPs are implemented first that will alter peak stormflows, design the monitoring program to track changes in hydrology. After the needed hydrologic restoration is achieved, monitoring can be redirected to track expected changes in channel morphology. Once changes in channel morphology are documented, monitoring can then focus on assessment of habitat and biological community response. Response of stream hydrology is likely to be quicker than restoration of stream biota and would therefore be a valuable—and more prompt—indicator of progress.

Conclusions

Lag time between implementation of management practices on the land and water quality response is an unfortunate fact of life in watershed management. Unless recognized and dealt with, long lag time will frequently confound our ability to successfully document improved water quality resulting from treatment of NPS and may discourage vital restoration efforts. While ongoing and future research may provide us with better tools to predict and account for lag time, it is essential that watershed monitoring programs today recognize and grapple with this issue. Useful approaches to deal with the inevitable lag between implementation of management practices and water quality response include educating stakeholders on reasonable expectations, appropriately characterizing the watershed and pollutant delivery processes; considering lag time in selection, siting, and monitoring of management measures; selection of appropriate indicators with which to assess progress; and designing effective monitoring programs to detect water quality response and document effectiveness.

References

Bester, M.L., E.O. Frind, J.W. Molson, and D.L. Rudolph. 2006. Numerical investigation of road salt impact on an urban wellfield. Ground Water 44:165–175.

Bratton, J.F., J.K. Böhlke, F.M. Manheim, and D.E. Krantz. 2004. Submarine ground water in Delmarva Peninsula coastal bays: Ages and nutrients. Ground Water 42:1021–1034.

Buzzards Bay National Estuary Program. 2008. Turning the tide on shellfish bed closures in Buzzards Bay during the 1990s. Available at http://www.buzzardsbay.org/shellclssuccess.htm (verified 19 Oct. 2009).

Byappanahalli, M.N., R.L. Whitman, D.A. Shively, M.J. Sadowsky, and S. Ishii. 2006. Population structure, persistence, and seasonality of autochthonous *Escherichia coli* in temperate, coastal forest soil from a Great Lakes watershed. Environ. Microbiol. 8:504–513.

Chadwick, J.W., S.P. Canton, and R.L. Dent. 1986. Recovery of benthic invertebrate communities in Silver Bow Creek, Montana following improved metal mine wastewater treatment. Water Air Soil Pollut. 28:427–438.

Clark, J.J., and P.R. Wilcock. 2000. Effects of Land-use change on channel morphology in northeastern Puerto Rico. Geol. Soc. Am. Bull. 112:1763–1777.

Clausen, J.C. 2004. Jordan Cover Urban Watershed Section 319 National Monitoring Program Project. Annual Report. College of Agriculture and Natural Resources, Univ. of Connecticut, Storrs.

Clausen, J.C., and J. Spooner. 1993. Paired watershed study design. USEPA Publ. 841-F-93–009. U.S. Environ. Protection Agency, Washington, DC.

Crane, S.R., and J.A. Moore. 1986. Modeling enteric bacterial die-off: A review. Water Air Soil Pollut. 27:411–439.

Cravotta, C.A., III. 2009. Abandoned mine drainage in the Swatara Creek Basin, Southern Anthracite Coalfield, Pennsylvania, USA: 1. Streamwater-quality trends coinciding with the return of fish. Mine Water Environ. (in press).

Dettmann, E.H. 2001. Effect of water residence time on annual export and denitrification of nitrogen in estuaries: A model analysis. Estuaries 24:481–490.

Druschel, G., A. Hartmann, R. Lomonaco, and K. Oldrid. 2005. Determination of sediment phosphorus concentrations in St. Albans Bay, Lake Champlain: Assessment of internal loading and seasonal variations of phosphorus sediment-water column cycling. Final Report prepared for Vermont Agency of Natural Resources, Dep. of Geology, Univ. of Vermont, Burlington.

Gale, J.A., D.E. Line, D.L. Osmond, S.W. Coffey, J. Spooner, J.A. Arnold, T.J. Hoban, and R.C. Wimberly. 1993. Evaluation of the experimental rural clean water program. EPA-841-R-93–005. Natl. Water Quality Evaluation Project, NCSU Water Quality Group, Biological and Agric. Engineering Dep., North Carolina State Univ., Raleigh.

Galeone, D.G. 2005. Pequea and Mill Creek Watersheds Section 319 NMP Project: Effects of Streambank Fencing on Surface-water Quality. NWQEP Notes 118: August, 2005. North Carolina State Univ., Raleigh.

Giroux, M., and R. Royer. 2007. Long term effects of phosphate applications on yields, evolution of P soil test, saturation, and solubility in two very rich soils. Agrosolutions 18:17–24. Inst. de recherche et de développement en agroenvironnement (IRDA), Québec, Canada. (In French, with English abstract.)

Goodman, J., J.M. Collins, and K.B. Rapp. 1992. Nitrate and pesticide occurrence in shallow groundwater during the Oakwood Lakes-Poinsett RCWP project. p. 33–45. *In* The National Rural Clean Water Program Symp., September 1992. Orlando, FL. ORD EPA/625/R-92/006. USEPA, Washington, DC.

Gunsalus, B., E.G. Flaig, and G. Ritter. 1992. Effectiveness of agricultural best management practices implemented in the Taylor Creek/Nubbin Slough Watershed and the Lower Kissimmee River Basin. p. 161–171. *In* The National Rural Clean Water Program Symp., September 1992. Orlando, FL. ORD EPA/625/R-92/006. USEPA, Washington, DC.

Halvorson, A.D., and C.A. Reule. 1980. Alfalfa for hydrologic control of saline seeps. Soil Sci. Soc. Am. J. 44:370–374.

Jaworski, N. 1990. Retrospective of the water quality issues of the upper Potomac estuary. Aquat. Sci. 3:11–40.

Jokela, W.E., J.C. Clausen, D.W. Meals, and A.N. Sharpley. 2004. Effectiveness of agricultural best management practices in reducing phosphorus loading to Lake Champlain. p. 39–52. *In* T. Manley et al. (ed.) Lake Champlain: Partnership and research in the new millennium. Kluwer Academic/Plenum Publ., New York.

Kemp, W.M., W.R. Boynton, J.E. Adolf, D.F. Boesch, W.C. Boicourt, G. Brush, J.C. Cornwell, T.R. Fisher, P.M. Glibert, J.D. Hagy, L.W. Harding, E.D. Houde, D.G. Kimmel, W.D. Miller, R.I.E. Newell, M.R. Roman, E.M. Smith, and J.C. Stevenson. 2005. Eutrophication of Chesapeake Bay: Historical trends and ecological interactions. Mar. Ecol. Prog. Ser. 303:1–29.

King, K.W., P.C. Smiley, Jr., B.J. Baker, and N.R. Fausey. 2008. Validation of paired watersheds for assessing conservation practices in the Upper Big Walnut Creek watershed, Ohio. J. Soil Water Conserv. 63:380–395.

Koerkle, E.H. 1992. Effects of nutrient management on surface water quality in a small watershed in Pennsylvania. p. 193–207. *In* The National Rural Clean Water Program Symposium, September 1992. Orlando, FL. ORD EPA/625/R-92/006. USEPA, Washington, DC.

Krantz, D.E., F.T. Manheim, J.F. Bratton, and D.J. Phelan. 2004. Hydrogeologic setting and ground-water flow beneath a section of Indian River Bay, Delaware. Ground Water 42:1035–1051.

Laflen, J.M., R. Lal, and S.A. El-Swaify. 1990. Soil erosion and a sustainable agricultura. p. 569–581. *In* C.A. Edwards et al. (ed.) Sustainable agriculture systems. Soil and Water Conserv. Soc., Ankeny, IA.

Lake Champlain Basin Program. 2008. State of the lake and ecosystem indicators report—2008. Available at http://www.lcbp.org/PDFs/SOL2008-web.pdf (verified 19 Oct. 2009). Lake Champlain Basin Program, Grand Isle, VT.

LakeNet. 2009. Lake profile: Champlain. Available at http://www.worldlakes.org/lakedetails.asp?lakeid = 8518 (verified 19 Oct. 2009). LakeNet Secretariat, Annapolis, MD.

Larsen, D.P., J. Van Sickle, K.W. Malueg, and P.D. Smith. 1979. The effect of wastewater phosphorus removal on Shagawa Lake, Minnesota: Phosphorus supplies, lake phosphorus, and chlorophyll a. Water Res. 13:1259–1272.

Line, D.E., and G.D. Jennings. 2002. Long Creek watershed nonpoint source water quality monitoring project–Final report. North Carolina State Univ., Raleigh.

Marshall, D.W., A.H. Fayram, J.C. Panuska, J. Baumann, and J. Hennessey. 2008. Positive effects of agricultural land use changes on coldwater fish communities in Southwest Wisconsin streams. N. Am. J. Fish. Manage. 28:944–953.

Marutani, T., M. Kasai, L.M. Reid, and N.A. Trustrum. 1999. Influence of storm-related sediment storage on the sediment delivery from tributary catchments in the Upper Waipaoa River, New Zealand. Earth Surf. Processes Landforms 24:881–896.

McCollum, R.E. 1991. Buildup and decline of soil phosphorus: 30-Year trends on a Typic Umprabuult. Agron. J. 83:77–85.

Meals, D.W. 1992. Water quality trends in the St. Albans Bay, Vermont watershed following RCWP land treatment. p. 47–58. *In* The National Rural Clean Water Program Symp., September 1992. Orlando, FL. ORD EPA/625/R-92/006. USEPA, Washington, DC.

Meals, D.W. 1993. Assessing nonpoint source phosphorus control in the LaPlatte River watershed. Lake Reservoir Manage. 7:197–207.

Meals, D.W. 1996. Watershed-scale response to agricultural diffuse pollution control programs in Vermont, USA. Water Sci. Technol. 33:197–204.

Meals, D.W. 2001. Water quality response to riparian restoration in an agricultural watershed in Vermont, USA. Water Sci. Technol. 43:175–182.

Miller, H.R., P.L. Brown, J.J. Donovan, R.N. Bergatino, J.L. Sondregger, and F.A. Schmidt. 1981. Saline seep development and control in the North American Great Plains—Hydrogeological aspects. Agric. Water Manage. 4:115–141.

Newbold, J.D., J.W. Elwood, R.V. O'Neil, and W. Van Winkle. 1981. Measuring nutrient spiralling in streams. Can. J. Fish. Aquat. Sci. 38:860–863.

Newbold, J.D., S. Herbert, B.W. Sweeney, and P. Kiry. 2008. Water quality functions of a 15-year-old riparian forest buffer system. p. 1–7. *In* Riparian ecosystems and buffers: Working at the water's edge. AWRA Summer Specialty Conf. Am. Water Resources Assoc., Virginia Beach, VA.

Newson, J. 2007. Measurement and modeling of sediment transport in a northern Idaho stream. Masters thesis. Biological and Agric. Eng., College of Graduate Studies, Univ. of Idaho, Moscow.

Nixon, S.W., J.W. Ammerman, L.P. Atkinson, V.M. Berounsky, G. Billen, W.C. Boicourt, W.R. Boynton, T.M. Church, D.M. Ditoro, R. Elmgren, J.H. Garber, A.E. Giblin, R.A. Jahnke, N.J.P. Owens, M.E.Q. Pilson, and S.P. Seitzinger. 1996. The fate of nitrogen and phosphorus at the land-sea margin of the North Atlantic Ocean. Biogeochemistry 35:141–180.

Owens, L.B., M.J. Shipitalo, and J.V. Bonta. 2008. Water quality response times to pasture management changes in small and large watersheds. J. Soil Water Conserv. 63:292–299.

Phillips, S.W., and B.D. Lindsey. 2003. The influence of ground water on nitrogen delivery to the Chesapeake Bay. USGS Fact Sheet FS-091–03. Available at http://md.water.usgs.gov/publications/fs-091–03/index.html (verified 19 Oct. 2009). U.S. Geological Surv., MD-DE-DC Water Sci. Ctr., Baltimore, MD.

Pote, D.H., T.C. Daniel, A.N. Sharpley, P.A. Moore, D.R. Edwards, and D.J. Nichols. 1996. Relating extractable soil phosphorus to phosphorus losses in runoff. Soil Sci. Soc. Am. J. 60:855–859.

Quinn, F.H. 1992. Hydraulic residence times for the Laurentian Great Lakes. J. Great Lakes Res. 18:22–28.

Rao, P.S.C., A.G. Hornsby, and R.E. Jessup. 1985. Indices for ranking the potential for pesticide contamination of groundwater. Proc. Soil Crop Sci. Soc. Fla. 44:1–8.

Richards, R.P., and G.L. Grabow. 2003. Detecting reductions in sediment loads associated with Ohio's conservation reserve enhancement program. J. Am. Water Resour. Assoc. 39:1261–1268.

Roseboom, D.P., T.E. Hill, J. Beardsley, J.A. Rodsater, and L. Duong. 1996. Waukegan River National Monitoring Program, biological and physical monitoring of Waukegan River restoration efforts in biotechnical bank protection and pool/riffle creation, 1996. ISWS CR-629:1998 Illinois State Water Survey, Champaign.

Schilling, K.E., and J. Spooner. 2006. Effects of watershed-scale land use change on stream nitrate concentrations. J. Environ. Qual. 35:2132–2145.

Schilling, K.E., and C.F. Wolter. 2007. A GIS-based groundwater travel time model to evaluate stream nitrate concentration reductions from land use change. Environ. Geol. 53:433–443.

Scientific and Technical Advisory Committee, Chesapeake Bay Program. 2005. Understanding "lag times" affecting the improvement of water quality in the Chesapeake Bay. STAC Publ. 05–001. Available at http://www.chesapeake.org/stac/Pubs/LagTimeReport.pdf (verified 19 Oct. 2009). Chesapeake Bay Program, Edgewater, MD.

Seo, D., and R.P. Canale. 1999. Analysis of sediment characteristics and total phosphorus models for Shagawa Lake. J. Environ. Eng. 125:346–350.

Sharpley, A.N., S. Herron, and T. Daniel. 2007. Overcoming the challenges of phosphorus-based management in poultry farming. J. Soil Water Conserv. 62:375–389.

Sherer, B.M., J.R. Miner, J.A. Moore, and J.C. Buckhouse. 1992. Indicator bacterial survival in stream sediments. J. Environ. Qual. 21:591–595.

Sims, J.T., A.C. Edwards, O.F. Schoumans, and R.R. Simard. 2000. Integrating soil phosphorus testing into environmentally based agricultural management practices. J. Environ. Qual. 29:60–71.

Spooner, J., C.J. Jamieson, R.P. Maas, and M.D. Smolen. 1987. Determining statistically significant changes in water quality pollutant concentrations. Lake Reservoir Manage. 3:195–201.

Spooner, J., L.A. Szpir, D.E. Line, D.L. Osmond, D.W. Meals, and G.L. Grabow. 2008. 2008 Summary Report: Section 319 National Monitoring Program Projects, National Nonpoint Source Watershed Project Studies, NCSU Water Quality Group, Biological and Agricultural Engineering Dep., North Carolina State Univ., Raleigh.

Tomer, M.D., and M.R. Burkart. 2003. Long-term effects of nitrogen fertilizer use on ground water nitrate in two small watersheds. J. Environ. Qual. 32:2158–2171.

USEPA. 2005. Handbook for developing watershed plans to restore and protect our waters. EPA 841-B-05–005. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Virginia Department of Conservation and Recreation. 1996. Alternative watering systems for livestock-The Middle Fork Holston River builds on success. Available at http://www.epa.gov/owow/NPS/Section319II/VA.html (verified 19 Oct. 2009). USEPA Office of Wetlands, Oceans, and Watersheds, Washington, DC.

Wang, D., S.N. Levine, D.W. Meals, J.P. Hoffmann, J.C. Drake, and E.A. Cassell. 1999. Importance of in-stream nutrient storage to phosphorus export from a rural, eutrophic river in Vermont, USA. p. 205–223. *In* T.O. Manley and P.L. Manley (ed.) Lake Champlain in transition: From research toward restoration. Water Science and Application, Vol. 1, American Geophysical Union, Washington, DC.

Weston Solutions, Inc. 2005. Monitoring program recommendations for report of waste discharge. Report prepared for San Diego County Copermittees, County of San Diego, Watershed Protection, Dep. of Public Works. Available at http://www.westonsolutions.com/services/environmental_solutions/water_resources/index.htm (verified 19 Oct. 2009). Weston Solutions, Inc., Carlsbad, CA.

White, W.P., J. Beardsley, J.A. Rodsater, L. Duong, and T.E. Hill. 2003. Biological and physical monitoring of Waukegan River restoration efforts in biotechnical bank protection and pool/riffle creation: Annual data report, May 31, 2003. Illinois Environ. Protection Agency, Springfield, IL.

Whitney, L., and R. Hafele. 2006. Stream restoration and fish in Oregon's Upper Grand Ronde river system. NWQEP Notes. Available at http://www.bae.ncsu.edu/programs/extension/wqg/issues/notes123.pdf (verified 19 Oct. 2009). North Carolina State Univ. Coop. Ext., Raleigh.

Zhang, T.Q., A.F. MacKenzie, B.C. Liang, and C.F. Drury. 2004. Soil test phosphorus and phosphorus fractions with long-term phosphorus addition and depletion. Soil Sci. Soc. Am. J. 68:519–528.