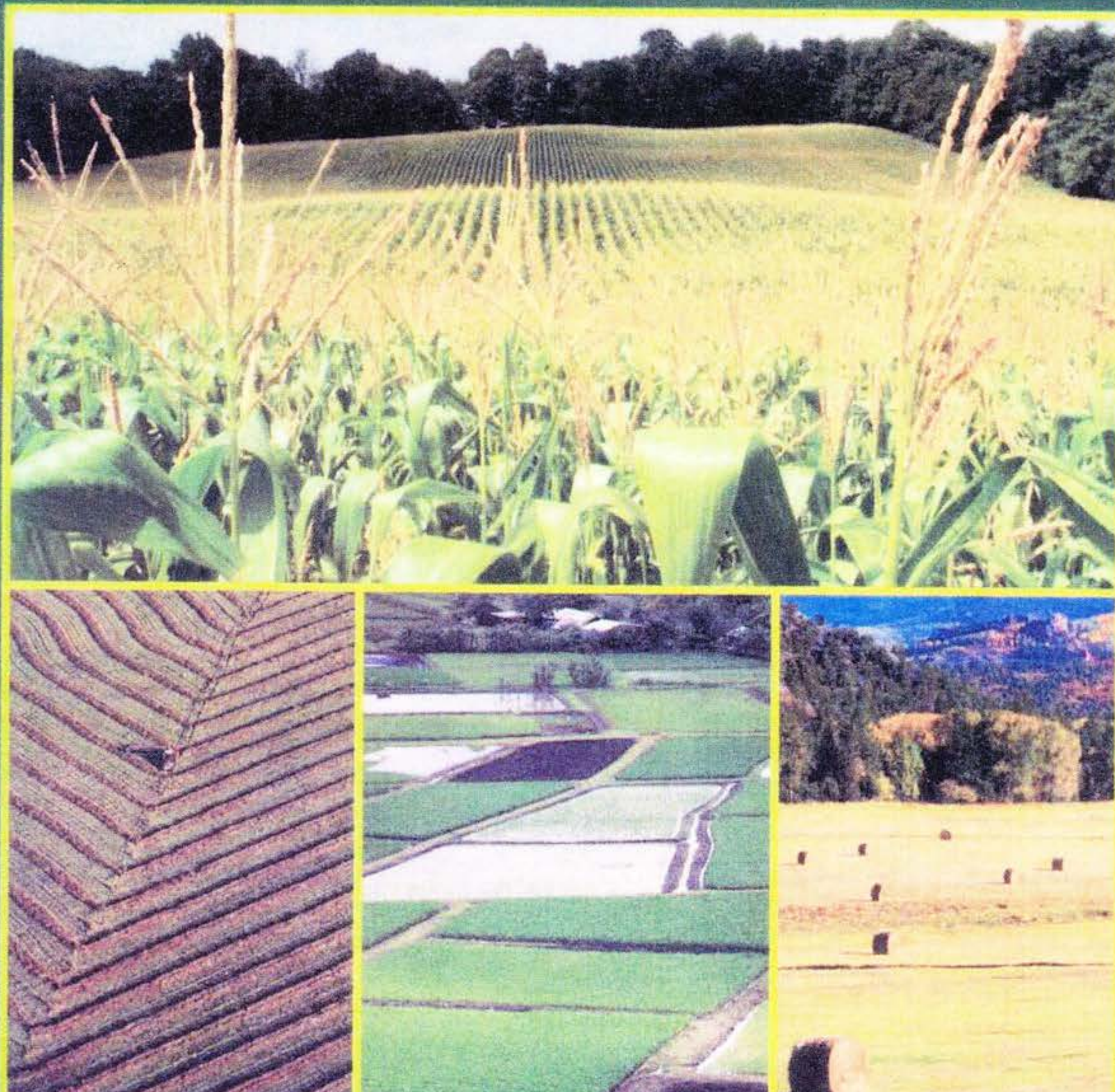
Managing Agricultural Landscapes for Environmental Quality

Strengthening the Science Base

Max Schnepf and Craig Cox, Editors



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The Soil and Water Conservation Society (SWCS) is a nonprofit scientific and educational organization that serves as an advocate for natural resource professionals and for science-based conservation policy. SWCS fosters the science and art of soil, water, and environmental management on working lands to achieve sustainability. SWCS members promote and practice an ethic that recognizes the interdependence of people and their environment.

The science of targeting within landscapes and watersheds to improve conservation effectiveness

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Certain portions of the landscape are especially sensitive to human activities or strategically located to mitigate the environmentally detrimental impacts of human activities. Targeted land management is thus defined here as the focusing of preservation, conservation, or other practices on those specific portions of the landscape (or at particular times) where (and when) they will have the greatest benefits at the lowest economic costs (assuming that conservation practices actually reduce net economic return).

Motivation and need for targeted management are increasingly acute as society's pressures on and valuing of natural resources simultaneously increase. Ideally, targeted land management allows conservation funds to improve environmental quality most effectively and balance a complex and shifting mosaic of land uses that vie for space and differ in the economic returns they provide. In the case of land conservation programs, this funding is often needed to compensate landowners for adopting environmentally friendly practices that involve foregoing some or all of the economic returns to land.

The "targeting" concept evolves around improved scientific understanding of how ecological, hydrological, and geomorphological processes and economic costs of land management are distributed and interact across the landscape and over time. These advances in knowledge of ecological and earth-system connectivity have paralleled improvements in geographic information systems (GIS), remote sensing, and other methods of gathering, compiling, and analyzing geospatial data. Because of the increasingly complex suite of management objectives, effective implementation requires scientifically based planning tools, guidelines, or strategies to match practices and locations with optimal effectiveness.

To help focus this discussion, we primarily consider spatial rather than temporal aspects of targeting, for which the reader is referred to other sources (e.g., Edwards et al., 1992; Randall and Mulla, 2001; Dinnes et al., 2002). Here then, targeting is usually the process of identifying priority locations for implementing conservation or habitat improvement practices. Ideally, targeted management incorporates the combined or integrated effect of many individually targeted practices (e.g., Vieth et al., 2003, 2004). Similarly, targeted landscape management often balances multiple objectives, usually protecting water quality, soil health, terrestrial or aquatic habitat, and/or public safety while simultaneously recognizing economic and other management issues that may be ownership-specific. Planners and decision-makers may choose among a variety of balancing criteria, such as maximizing environmental benefits achieved, minimizing economic costs of achieving environmental benefits, and maximizing environmental benefits subject to some cost or land-area constraint, among others. The choice of objectives is important to implementing a targeted management plan because the correlation between the distribution of environmental benefits and economic costs of land parcels within a region plays an important role in deciding which land parcels get targeted (Babcock et al., 1996, 1997). For purposes of this discussion, targeting seeks to optimize the beneficial effects of conservation practices on natural resources with a given conservation budget and thereby maximize the returns on economic investments in conservation.

Targeting as a process is not restricted to any particular scale of space or time. The scale at which we target land management depends upon the interplay among a number of factors, including specific management objectives, relevant ecological or earth-system processes, the precision at which management decisions can be implemented, the resolution of available data, and potential sociopolitical constraints. One can find examples of targeted management across a wide range of scales, from very large areas (e.g., preservation of terrestrial habitats) and whole watersheds (e.g., U.S. Environmental Protection Agency hi-priority watersheds) to individually owned parcels (e.g., nutrient management plans for farms) or smaller areas (e.g., riparian areas or stormwater infiltration basins).

The case for targeting

There are three compelling reasons for promoting targeted land management to meet resource conservation or preservation goals. First, agriculture is a "landscape" enterprise, of which the fields in agricultural production are only a part. Landscapes are, by definition, heterogeneous. If all places in a landscape were the same or if we could consider all places (agricultural fields, streams, watersheds) independent of their surroundings, there would be no need for targeting. But the context of agricultural fields (even seemingly homogenous ones) in a spatially varied landscape creates the need for and the opportunity to use targeting to improve the effectiveness of land use and conservation management.

Second, there is irrefutable scientific evidence that some locations in the landscape have a high pollutant-generating potential (sensitive sites), can function effectively to intercept and treat pollutants, and/or have features that comprise critical habitat for wildlife. By understanding how such processes or properties vary spatially, efforts to mitigate their detrimental effects or enhance their beneficial consequences can be applied more precisely.

Third, the economic costs of conservation practices also differ across locations. Accordingly, targeting allows cost-effective conservation efforts to focus funds for resource conservation in locations where consequences, such as pollution-reduction potential, are high and costs are low (Johansson and Randall, 2003).

It is not possible to apply conservation treatments across the entire landscape. Ecosystems cannot all be entirely reconstructed, and we must accept that human activities impact soil, water, and habitat resources. Therefore, methods are needed to identify and protect areas that are sensitive (most prone to environmental damage), able to effectively mitigate pollution, and/or provide critical habitat. The decision to adopt a conservation practice to protect the environment is most effective if it is a voluntary decision by a landowner, rather than one mandated by government. Ideally, such decisions would be based on increased social consciousness toward environmental protection through education and technical assistance that encourages landowners to adopt conservation practices. Moral suasion and efforts to raise this kind of consciousness, however, have an insignificant effect in promoting adoption of conservation practices that are unprofitable unless the landowner's immediate environment or health is at risk. In contrast, economic incentives through cost-sharing do have a significant impact on conservation behavior (Batie and Ervin, 1999). Designing market-based incentives for landowners to adopt conservation practices, therefore, has potential for improving participation in environmental protection.

The key questions in designing and implementing conservation policies that employ targeting, then, are (1) how do we develop methods that identify specific targeting criteria for each resource, especially in the context of multiple-criteria objectives and (2) how do we translate these criteria into conservation planning tools/instrumentation that can be developed at regional or watershed scales and applied at the farm-field (ownership) scale in ways that are cost-effective across multiple agricultural producers?

Targeted management: An evolving concept

Clearly, there are historical precedents for targeting conservation practices. Indeed, the National Park System is a form of targeted environmental protection. Also, agricultural producers intuitively understand targeting and have, for decades, routinely timed fertilizer applications to maximize crop uptake (Dinnes et al., 2002) and, more recently, utilized precision agricultural technologies to focus fertilizer applications on parts of the landscape where crop nutrient utilization will be maximized (Rejesus and Hornbaker, 1999; Kaspar et al., 2003).

Since the 1930s, U.S. Department of Agriculture (USDA) programs and activities have encouraged farmers to establish conservation practices on environmentally sensitive land. While primarily based on localized knowledge up through the 1970s (Heimlich, 2003), the 1985 farm bill and USDA's Conservation Reserve Program (CRP) more clearly defined areas that should be targeted to protect wetlands and reduce soil erosion. Specifically, USDA's universal soil loss equation, in conjunction with site-specific soil survey information, was used to define quantitatively highly erodible land (Benbrook, 1988) and a federal interagency definition of "wetland" was developed with hydric soils as a key indicator (Heimlich, 1994). More recently there has been an important, implicit shift from on-site impacts to down-system influences, for example, water quality impairment attributed to up-slope land management.

Concerns continue to broaden beyond soil erosion (highly erodible land) and water storage (wetlands) to include water quality, air quality, and wildlife. The number of governmental programs has expanded, and eligibility criteria for conservation treatments have become more specific. For example, the environmental benefits index developed for evaluating and ranking land parcels offered for CRP enrollment includes a detailed evaluation of potential for wildlife habitat and water quality improvement, in addition to protection of highly erodible soils (U.S. Department of Agriculture-Farm Service Agency, 1999) and the soil rental rate of each parcel. This index is based on landscape information, including highly erodible soils, proximity to water and wetlands, and location of state-designated critical areas for water quality (e.g., Clean Water Act Section 303d listed waters) or wildlife (e.g., habitat areas for species listed under the Endangered Species Act). These indices are typically used to rank conservation proposals from pools of voluntarily submitted applications to prioritize incentive payments. Despite such use of these indices, increasingly complex eligibility determinations still must be implemented on the basis of professional judgment and interpretations of general soil and landscape position rather than on detailed, robust scientific understanding of landscape functions, processes, and quantitative relationships.

Over the past decade, local planning efforts have become important driving forces behind targeted landscape management, especially with respect to watershed management for water quality and aquatic habitat protection (e.g., Walter and Walter, 1999; House, 2000; hundreds of plans are not widely publicized). Local officials are generally eager for sound scientific bases to prioritize water quality improvement projects. One high-profile example is the New York City (NYC) Agricultural Watershed Protection Project (a.k.a., Watershed Agriculture Program/WAP; Walter and Walter, 1999), which is directed by a Watershed Agriculture Council (WAC) largely made up of local producers and landowners who partner with regional research universities to develop the underpinning science for targeted management practices.

Some states have developed formal legislation to promote these types of grassroots efforts, such as Washington State's 1998 Watershed Planning Act, which states "...the local development of watershed plans for managing water resources... is vital to both state and local interests." Indeed, this type of localization of management allows scientists to develop targeted management strategies that match local or regional conditions and controlling processes in ways that can be more difficult under broader, usually nationally mandated efforts. Explicitly included in the NYC watershed program was a requirement to "test, demonstrate, and evaluate the scientific...criteria developed for whole farm planning (New York State Water Resources Institute, 1992)," most of the components of which were various targeted best management practices (BMPs) at farm and field scales for controlling nonpoint-source pollution in NYC water-supply reservoirs (e.g., Walter et al., 2000, 2001).

Although there has been no formal analysis, one common, implicit theme among successful watershed projects is a close interaction between landowners, planners, and/or researchers (e.g., Meals, 2001; Wang et al., 2002; Dietz et al., 2004; Bishop et al., 2005; Lovegreen et al., 2006). Those types of locally driven efforts also facilitate accommodating the differences in priorities, concerns, and objectives among individual landowners, local planners, and broader state and national governments.

It appears that current trends in targeted land management are a natural part of the progression of environmental conservation history. Hopefully, the shift over the past decade or so toward locally initiated conservation and protection efforts will continue to redefine the role of top-down policies. For example, the Conservation Reserve Enhancement Program (CREP), a federal-state partnership program initiated in 1996, has helped regionalize cost-effective targeting of high-priority, environmentally sensitive land based on quantitatively defined environmental goals (U.S. Department of Agriculture-Farm Service Agency, 2003), although it could be improved to better engage grassroots activities. As discussed earlier, increased local control also potentially facilitates closer ties between planners or researchers and landowners, ensuring better application of scientific concepts to targeting efforts.

Although targeting allows planners to realize the continuum of objectives and natural processes distributed across the globe, we focus here on water quality and terrestrial habitat protection for wildlife, two prominent objectives that illustrate the breadth of science, technology, and future challenges and opportunities of targeted land management.

Water quality protection and enhancement

The state of the scientific support for targeting

An important, recent shift in targeting of conservation practices is an improved understanding of how individual parts of the landscape are interconnected. Early water quality protection efforts involved little more than a repackaging of soil conservation practices (Walter et al., 1979; Walter et al., 2000). Although our efforts to gain knowledge of critical processes and their scales will surely continue to challenge us, scientific advances during the past 30 years have provided ample justification for targeted water quality protection practices beyond those currently employed as BMPs. In particular, there have been substantial advances in understanding how hydrological processes are distributed across the landscape (e.g., Hewlett and Hibbert, 1967; Dunne and Black, 1970a,b; O'Loughlin, 1981; Moore et al., 1991; Lyon et al., 2006a,c) and how those processes are influenced by soil variability (e.g., Thompson et al., 1997; Gessler et al., 2000; Western et al., 2004) and correlate with crop productivity (e.g., Kaspar et al., 2003; Kravchenko and Bullock, 2000).

Targeting of conservation practices is increasingly based on those advances in distributed hydrology and sound scientific understandings of relevant ecological and/or transport processes. Recent scientific, paired-watershed evaluations (Loftis et al., 2001), although somewhat nonspecific, suggest that a targeted approach can effectively improve water quality and aquatic health, especially practices that protect riparian areas (e.g., Meals, 2001; Wang et al., 2002; Dietz et al., 2004; Bishop et al., 2005). Studies to evaluate rigorously the effectiveness of targeted versus other nontargeted conservation implementations have not been conducted, however, nor do we have a good understanding of the system-wide impacts of specific implementations and/or maintenance of particular practices and combinations of practices. In lieu of good field evaluations, we have relied (perhaps too much) on model results to demonstrate the effectiveness of targeted management (e.g., Walter et al., 2001; Veith et al., 2004; Mankin et al., 2005). More reliable evaluations will require innovatively designed, long-term, and/or paired-watershed studies implemented at the appropriate scale. Furthermore, linkages between conservation practices and water quality response in large watersheds must bridge the scales of implementation (field) and measurement (watershed) or risk being experimentally flawed (see Gardner et al., 2001).

This bridging of scales is challenging, but the risk of ignoring this challenge is clearly illustrated, for example, by the recognition that soil conservation practices have not always resulted in anticipated reductions in stream sediment loads (e.g., Trimble, 1999a,b; Trimble and Crosson, 2000a,b). This is because of the widely employed, simplistic assumption that the amount of sediment generated by a land parcel and the amount reaching a water body depend only upon a fixed proportion or the distance of the parcel from the water body (e.g., Khanna and Farnsworth, 2005). So, although the scientific rationale for many (often targeted) soil conservation practices is sound and those practices do arrest excessive soil loss, the intended benefits for rivers have not been recognized. That is because we did not fully understand or consider the integration of practices, processes, and expected environmental benefits and thus failed to select appropriate experimental designs and monitoring scales. Experiments and monitoring to demonstrate water quality benefits from targeted practices need be designed to consider the landscape position of land parcels and consider how processes are distributed along flow paths; in this example, upland erosion, redeposited, and channel-bed erosion (Nagle and Ritchie, 1999, 2004; Nagle et al., 2006). Interestingly, Khanna et al. (2003) showed that retiring particular land parcels without considering the management decisions of surrounding parcels considerably underestimates the resulting sediment loads to streams, at much higher costs than targeted practices that consider land parcels in their hydrologic context. This soil erosion, stream sediment example also demonstrates the risk of relying too much on models to make predictions, that is, our overly simplistic assumptions of how parcel- or plot-scale erosion is linked to sediment delivery to streams were built into the models, which gave poor estimates of how targeted soil-conservation practices would improve stream water quality (e.g., Trimble, 1999b; Trimble and Crosson, 2000b).

In summary, the scientific rationale for many targeted practices is good. Indeed, the past decade has seen a notable shift toward targeted management strategies using process-based scientific understanding of these practices. Subsequent studies, although too few, suggest that the implementation of those practices is having positive impacts on environmental quality (e.g., Meals, 2001; Wang et al., 2002; Dietz et al., 2004; Bishop et al., 2005; Lovegreen et al., 2006). But we must continue to improve our understanding of how processes are integrated over whole systems at different scales, which will ultimately allow us to better evaluate targeted management successes and failures in ways that illuminate knowledge gaps and meaningfully direct future progress. It is important to remember, however, that the scientific bases for targeting conservation practices need to interface ultimately with planners and land managers in ways that combine local/regional knowledge and objectives to develop better and more economical environmental protection practices. In essence, an improved scientific basis for targeting of practices is allowing planners to move increasingly away from "one size (scale) fits all" management approaches. We comment further on the issue of scaling targeting in the concluding section of this paper.

The state of targeting technology

Tools for targeting biogeo processes. Technological advances in monitoring and managing increasingly complex, spatially distributed information and in precisely locating landscape positions have enabled revolutionary advances in targeted land management capabilities. While we do not want to diminish the contributions these technological advances have made to better targeted water quality protection and enhancement, we think it is arguably more important to recognize that many targeting approaches to protect water quality are based on decades-old data (e.g., soil surveys), methods (e.g., soil phosphorus tests), and models (e.g., curve number runoff model, U.S. Department of Agriculture-Soil Conservation Service, 1972), which have been recycled through user-friendly GIS in ways that were neither anticipated nor intended by their originators. These "old" tools and technologies generally worked well for their intended purposes, but we cannot expect them to be uniformly applicable for effective targeting of water quality management across all regions or large river basins. Scientifically, we should be able to state our assumptions about the utility of each data resource used for targeting, and we must understand the likelihood of those assumptions being invalid and implications of employing them erroneously. In short, if targeting is to become part of policy one day, then at some point targeting criteria may need to be defended in court.

To illustrate this issue, one pervasive example is that among modelers and environmental protection practitioners there may be an implicit assumption that electronic geospatial data all have similar levels of precision or scientific reliability. While some data, like the newest digital elevation data (e.g., LIDAR), are very detailed and their errors well documented, others, especially soils data, are generally digitized forms of hand-drawn maps. Of course, soil scientists mapping North America's agricultural soils, largely during the mid-20th century, understood that soils are expressed as a landscape continuum, but mapping capabilities at the time required them to delineate general groups of soil characteristics by polygons on a map. While this traditionally discrete format for representing soil properties has been historically convenient for soil conservation targeting (e.g., identifying highly erodible land) or wetland identification (as indicated by hydric soils), these same data are incorporated into increasingly sophisticated analyses and mechanistic models without questions about their accuracy, precision, or variability, which can be significant from region to region. Despite this, some process-based hydrologic models perform well using these soils data (e.g., Frankenberger et al., 1999; Mehta et al., 2004; Gerard-Marchant et al., 2006; Schneiderman et al., 2006; Easton et al., 2006), suggesting that the current level of precision of soil survey data is adequate for targeting applications in at least some regions.

Similarly, there is an intriguing misconception that currently used water quality models, such as the soil and water assessment tool (SWAT) (e.g., Arnold et al., 1993), agricultural nonpoint-source pollution model (AGNPS) (Young et al., 1989), and the generalized watershed loading function (GWLF) (Haith and Shoemaker, 1987), provide highly targeted management insights, presumably because they use geospatially referenced data and the resulting model output can be presented as a digital map. Unfortunately, these models typically rely on the USDA Soil Conservation Service [now the Natural Resources Conservation Service (NRCS)] curve number equation (e.g., U.S. Department of Agriculture-Soil Conservation Service, 1972) to predict runoff. Although tables are available that link runoff potential (i.e., curve numbers) to specific land uses, there is little reliable scientific basis for these indices of runoff potential, especially for small storms (Walter and Shaw, 2005). Indeed, the model's creator, Victor Mockus, justified his model largely "on grounds that it produces rainfall–runoff curves of a type found on natural watersheds" (Rallison, 1980). So while the curve number method may provide a reasonable estimate of watershed-scale runoff, and, indeed, has been a useful engineering design tool for many decades, in most cases it cannot predict runoff from a specific location within a watershed. As currently used, then, water quality models may be reasonably effective at targeting priority watersheds, but are unlikely to target specific locations within a watershed for implementing BMPs or other water quality protection strategies (e.g., Garen and Moore, 2005). On a positive note, there have been substantial efforts to reconceptualize the traditional curve number equation in ways that are scientifically defensible for particular situations (e.g., Steenhuis et al., 1995; Gburek et al., 2002; Lyon et al., 2004; Scneiderman et al., 2006).

There are also several other terrain analyses that have potential applicability for improving the physical basis of water quality models with respect to specific runoff processes at hillslope scales (Moore et al., 1991; Bren, 1998; Gburek et al., 2002; Tomer et al., 2003; Agnew et al., 2006; Lyon et al., 2006b). Although more sophisticated process-based models may eventually provide a more scientifically robust hydrologic basis for water quality models, for the most part they are currently too cumbersome for practical use. In the foreseeable future it appears that institutional momentum will perpetuate the use of current water quality models and, hopefully, practitioners will adopt approaches for improving the scientific basis and decision-support capacities of these models.

We do not have space to discuss similar, potentially problematic reinventions of traditional data and approaches (for example, soil phosphorus tests originally developed to assess cropland productivity are now commonly used to assess phosphorus loading to streams). As with improvement in digital elevation data (e.g., LIDAR), it is likely that the accuracy and precision of other geospatial data will continue to improve as new environmental monitoring technologies mature. In the short term, it is important to recognize explicitly the gap between scientific advances and current technologies and tools for water quality "targeting." While the greater user-friendliness of these tools has improved their accessibility, it also risks facilitating their misuse. Thus, effective use of these tools could be improved by developing guidelines that explicitly address their technological limitations. That probably requires the professional judgment of specialists in combination with local knowledge.

Economic instruments to support targeting. Identification of land parcels that should be targeted for improved management to achieve environmental benefits at least cost can be achieved, at least in principle, by integrating an economic model with a water quality model, together with detailed GIS information on the characteristics of land parcels. Such integrated models typically assume that the economic costs of adopting conservation practices by a landowner are known and observable to a policy planner. They generally ignore the possibility of asymmetric information because the true cost of adopting a conservation practice is private information known only to the landowner. They also typically ignore the possibility of moral hazard: A landowner may adopt a conservation practice, but not make a full commitment to implementing it. Nevertheless, economic models that incorporate spatial heterogeneity in costs and physical processes and the implicit interdependencies of associated sediment-abatement benefits can improve policy planners' abilities to target conservation practices to reduce offsite pollutant loadings (Khanna et al., 2003).

Targeting, as referred to here, is defined normatively and from the perspective of a policy maker: It identifies land parcels on which conservation practices should be adopted to achieve environmental goals most cost-effectively. Because conservation efforts by landowners represent voluntary decisions in response to market-based incentives, policymakers need to design a "green payment" policy, such as subsidies for adoption of a conservation practice on cropland or a rental payment for retiring a land parcel from crop production, to provide incentives for landowners to adopt costly conservation practices on the targeted land parcels. Integrated, spatially explicit economic models can be used to design these per-acre green payments. In the presence of spatial heterogeneity in costs and environmental benefits, such payments may need to be parcel-specific in order to achieve conservation goals. For example, to achieve sediment-abatement goals through land retirement most cost-effectively, economic models show that the per-acre rental payments to landowners should vary with the location of the parcel relative to water bodies, the quality of the soil, the slope of the land parcel, and the soil erodibility index. To implement these site-specific rental payments, policymakers or planners need to know the relationships and parameters embedded in the integrated model. This is information- and skill-intensive. In practice, conservation programs tend instead to adopt second-best approaches to target land use change. For example, soil rental payments offered for enrollment in the CRP are soil-specific, but do not vary with the environmental benefits provided by that parcel. As another example, the Illinois CREP targets environmentally sensitive land by limiting eligibility for enrollment in the program to land parcels within a narrowly defined area in the Illinois River Basin. But it does not specify any mechanism to select 132,000 acres from the 7 million acres of heterogeneous land parcels in this area, which may ultimately raise program costs (Yang et al., 2004). Similar issues are prevalent with this program across the United States.

Challenges and future directions

What is needed to advance the targeting of water quality management and policies? Foremost, national targeting policies are needed that promote linkages between local, grassroots efforts and regional scientific researchers in ways that bridge science-application gaps. Interestingly, this need is echoed in one of the earliest discussions on targeting conservation practices by Maas et al. (1985), who emphasized that setting targeting criteria should take place within the context of watershed-specific planning. In other words, scientific knowledge must be adapted locally to set targeting criteria that are useful for site-specific decision-making; thus, human judgment is an important part of this process. For example, setback distances from water bodies must combine scientifically defensible criteria (e.g., Tomer et al., 2003, Walter et al., 2005) with local knowledge that is not necessarily applicable across broad regions. Indeed, this type of policy speaks directly to the problems associated with tensions between historical approaches and new technologies. Presumably, scientists are aware of these potential problems and can help local planners meaningfully interpret data and model results that may have "hidden" limitations, which might only be revealed through feedback from landowners or field reviews. Scientists are also uniquely qualified to recognize controlling processes and, thus, appropriate analytical approaches. For example, in some regions identifying hydrologically active areas for runoff generation (specifically, variable-source areas) is important to targeting water quality management practices. There are several scientifically defensible methods for identifying these areas based on topography (e.g., O'Loughlin, 1986; Vertessy et al., 1999; Mehta et al., 2004; Agnew et al., 2006), although these must be interpreted in the context of the planning objectives and the accuracy of available topographic data (Kuo et al., 1999; Tomer et al., 2003; Tomer and James, 2004), both of which can vary substantially from region to region.

A potential, though untested, dimension to this policy is that targeted management may be inherently more successful when scientists, planners, and landowners work closely together to establish linkages of trust. For example, Cornell researchers and the New York State Department of Environmental Protection worked for more than 10 years with an agricultural producer to target BMPs effectively in a small watershed in Delaware County and demonstrated substantial (about 40 percent) reductions in phosphorus and sediment loading with targeted land management (Bishop et al., 2005). Scientists and planners at the Bradford County, Pennsylvania Conservation District demonstrated similar pollutant reductions in a small watershed after more than 10 years of close cooperation with producers to develop appropriate, targeted practices (Lovegreen et al., 2006).

Of course, extensive monitoring allowed researchers to evaluate the success of those projects, but such monitoring is rare. Linking researchers, planners, and landowners encourages the implementation of monitoring systems that can provide scientifically valid assessments of the impacts of combinations of targeted practices on local- and wide-scale environmental impacts. Only in this way can experience with targeted management provide important feedback for improving future targeted management. Ideally, such efforts will implement distributed monitoring networks, in addition to measures at watershed outlets, to characterize how hydrologic and biogeochemical processes are distributed and interact across the landscape and in streams. Those processes interact to create "biogeochemical hotspots," parts of the landscape where pollutant or nutrient mobility or retention may be especially acute (McClain et al., 2003). Substantial background research supports the conclusion that biogeochemical processes are distributed or punctuated across the landscape (e.g., McClain et al., 2003; Welsh et al., 2005) and it is likely that some current targeting strategies could be more effective if they took these into account. One notable example is recent work showing that reclaiming incised urban streams reconnects the stream channel and riparian area in ways that increase denitrification (e.g., Groffman et al., 2002; Groffman and Crawford 2003). Improved understanding of how seasonal hydrologic cycles interact with microbial ecosystems reveals why wetlands and buffers act as nutrient (especially phosphorus) sources for some parts of the year and sinks during others (e.g., Dillaha et al., 1988; Carlyle and Hill 2001).

One important scientific challenge is determining the mechanics and transport roles of shallow, rapid, near-surface flows, such as those associated with lateral, preferential flowpaths (e.g., McDonnell, 2005). Subsurface preferential flow paths (Beven and German, 1982; Flury et al., 1994; Kung, 1990; Sidle et al., 2001) may be important within hillslopes (e.g., Wilson et al., 1990; Noguchi et al., 2001) and at watershed scales (e.g., Gatcher et al., 1998; Angier et al., 2005). There is good evidence that even compounds traditionally perceived as strongly adsorbed to soil, such as phosphorus, can move rapidly and deeply into the soil profile and then move laterally through natural preferential flowpaths (e.g., Gatcher et al., 1998) or artificial drainage systems (e.g., Geohring et al., 2001). Often, local landowners are the only sources of information about how and where artificial drainages are distributed through a landscape, reinforcing the need for scientists to work with local landowners to target land management practices.

Finally, creative research linking landscapes to streams through "tracer technologies" will enhance water quality targeting. These include genetic fingerprinting (e.g., Dombek et al., 2000;

Carson et al., 2001; Hartel et al., 2002, which may require genetic libraries on a watershed-specific basis; see Wiggins et al., 2003), natural isotopes (e.g., Christophersen et al., 1990; Genereux and Hooper, 1998; Kendall, 1998; McGlynn and McDonnell, 2003; Soulsby et al., 2003; Uhlenbrook and Hoeg, 2003), rare-earth elements (e.g., Lui et al., 2004; Kimoto et al., 2006), and anthropogenic sources of chlorofluorocarbons, sulfur-hexafluoride (e.g., Busenberg and Plummer, 2000; Browne and Gulden, 2005), and radionuclides (e.g., Wallbrink et al., 1999; Russell et al., 2001; Collins and Walling, 2002; Nagle et al., 2006). Innovative bio-nano-technological tracers (e.g., Mahler et al., 1998) can be used to identify pollutant sources and estimate the impacts of land use change on future trends in water quality. Through innovative combinations of these methods, researchers may be able to challenge the "nonpoint" concept, that is, the idea that we are incapable of identifying seemingly diffuse sources spread across the landscape. "Nonpoint-source pollution" may be more a label for our ignorance than a statement of reality. Ultimately, all pollutants have their sources in time and space.

Terrestrial habitat preservation for wildlife

The state of the scientific support for targeting

The previous section dealt with targeting to enhance water quality protection. The focus generally was on the diminishment of water quality by pollution, sedimentation, and the like. Stream systems and the factors affecting them are clearly multiscalar, and what happens upstream can have impacts at considerable distances from the sources (witness hypoxia in the Gulf of Mexico). Most water quality targeting, however, has been applied at local scales, often within individual landholdings or small watersheds. This is largely because governmental incentive programs for soil conservation and water quality improvement have been directed toward individual landowners. This is another reason why the engagement of landowners in targeting conservation efforts is so important.

For wildlife, the participation of local stakeholders is no less important. Because the objectives of wildlife conservation deal with species that have large ranges or migrate over vast distances, or with ecological systems whose dynamics are determined by regional as well as local factors, targeting efforts must be more explicitly multiscalar. But even though the scale of application of targeting may differ for different objectives, the importance of habitat in wildlife conservation carries across all scales.

Of course, the realization that species and ecological systems need habitat to survive and function was the foundation of natural history long before conservation and natural resource management emerged as recognizable disciplines, with their own professional societies, journals, and degree programs. Now, loss of habitat and fragmentation of what remains are widely considered to be primary threats to the persistence of populations and species (Forman and Godron, 1986; Dramstad et al., 1996; Wilcove et al., 1998). It would seem easy, then, to target "habitat" as the objective for protecting such populations and species. Indeed, the creation of nature reserves and protected areas is usually framed in terms of one habitat or another. Quite apart from debates about what "habitat" is or is not (e.g., Morrison and Hall, 2002), however, the recent emergence of landscape ecology, in tandem with the increasingly sophisticated use of spatially referenced GIS, has shown that this perspective by itself is inadequate and incomplete.

Similar to the earlier discussion regarding land parcels with respect to soil erosion, fragments of "habitat" are not independent areas floating in an inhospitable background matrix, but parts of a landscape mosaic of varied elements, with differing degrees of connectedness. Moreover, landscapes are not fixed and static in time, but rather dynamic, shifting, and changing as a result of natural disturbances and land use change (e.g., Turner et al., 1995; Dale and Haeuber, 2001; Theobald, 2005). And because the responses of different organisms to the landscape differ at different scales, as do different ecological processes, a "one size (scale) fits all" approach to protecting areas for conservation is unlikely to be as comprehensive or effective as one might wish. Collectively, these insights mean that targeting land management for the protection of species or ecological systems must be far more nuanced and much more challenging than simply setting aside areas of "habitat" and presuming the job is done. The situation is even more challenging for situations where habitat restoration or re-creation is attempted, although for much the same reasons that face habitat protection efforts.

The state of targeting technology

Tools for targeting habitat. Because "habitat" is so important to the conservation and management of species populations, the assessment of habitat requirements for organisms, particularly wildlife, has become a major arena of research activity. Habitat assessments have evolved through several phases: A qualitative phase of describing (qualitatively) habitat associations of species; a quantitative phase of simply correlating measures of individual habitat features with the abundance or density of a species, a preoccupation with increasingly sophisticated (and imponderable) multivariate analyses; and the use of computers to model habitat relationships in a predictive framework (Stauffer 2002). More recently, new statistical approaches, such as classification and regression trees (CART), artificial neural networks, and spatial autocorrelation analyses have brought new power to analyzing habitat patterns, and Akaike's information criterion procedures have provided ways to assess rigorously habitat model performance (Scott et al., 2002; Burnham and Anderson, 1998). Those tools, coupled with the use of remote sensing and GIS, have greatly enhanced our ability to understand how wildlife species respond to and occupy habitat.

At the same time, new approaches and technologies help land managers target areas in which to establish nature reserves. Rather than targeting places for protection based on vague, "pretty places," notions of conservation value, and opportunity, independently of one another, Australian scientists, in particular, advanced computer algorithms for reserve selection based on quantitative features of areas and their contributions to biodiversity protection, contingent on what is already protected ("complementarity"; Margules and Pressey, 2000; Margules, 2005). These approaches and the associated software (e.g., SITES and MARXAN; Groves, 2003) are being used by conservation planners to target and prioritize areas for conservation action based on multiple criteria. At the broad spatial scales of ecoregions (regions characterized by similar patterns of solar radiation and moisture and, consequently, by similar dominant plants and animals; Bailey, 1998), The Nature Conservancy (TNC) has been conducting ecoregional assessments to identify (target) a subset of areas that collectively represent the overall biological diversity of the ecoregion as a whole. This targeting process incorporates information on the distribution of characteristic and imperiled species, major plant communities, ecosystem types, and patterns of land ownership and management. The areas identified through this targeting process are then the foci for more intensive conservation efforts (e.g., land purchase for protection, conservation easements, cooperative agreements about land use) at more local scales (see Groves, 2003, for a summary of this approach).

Finally, landscape ecology is developing beyond the phase of documenting landscape pattern using GIS and deriving various measures of these patterns using software, such as FRAGSTATS (McGarigal et al., 2002), to incorporate information about how organisms respond to and move through complex landscape mosaics. The dispersal patterns of individuals in a population can be modeled and related to the continuities (e.g., corridors) or discontinuities (e.g., roads, housing developments) in a landscape to predict the probability that seemingly isolated habitats in a landscape may be functionally linked together (Wiens et al., 2002; Wiens, 2001). Formalized approaches of network analysis, diffusion, and percolation are being applied to create neutral models against which the patterns and dynamics of actual landscape linkages may be compared (e.g., Reiners and Driese, 2004; Jongman and Pungetti, 2004; With and King, 1997). These approaches are all aimed at integrating landscape structure with landscape function. How they can be employed to facilitate targeted land management, however, remains to be seen.

Economic instruments for ecosystem targeting. In the case of habitat preservation, most models for selecting sites for reserves to protect species have tended to focus on targeting the best locations for biological reserves to maximize the number of species that can be protected under a given budget constraint or under a restriction on the number or area of sites that can be conserved.

More recently, models have been developed that incorporate spatial criteria in reserve selection to identify the clustering of sites needed to enhance the long-term persistence of species and reduce fragmentation of preserved sites. The few studies that have included cost considerations in targeting habitat reserve sites have shown that budget-constrained site selection can result in more cost-effective conservation, especially where there is considerable spatial heterogeneity in land costs (Polasky et al., 2001). The aim, after all, is not just to be effective in targeting areas or actions for conservation, but to be most effective in applying the resources available to achieve the conservation goals. By applying return-on-investment approaches borrowed from economics to conservation targeting, the biological returns can be balanced against the economic costs, at least in theory (Wilson et al., 2006).

The targeting of sites for protecting species ultimately involves value judgments about whether preserving all species is equally important or if some species are more valuable than others. Often, reserve sites cannot all be protected immediately and excluded sites are threatened by development. The problem of targeting them involves choosing sites through time to include in a network of biological reserves for species conservation. This requires including both expected biodiversity benefits of sites and development risk. Costello and Polasky (2004) and Wilson et al. (2006) pointed out that the timing of targeting efforts is critical. Not only do conservation budgets available up front yield significantly greater biodiversity protection than the same investments delayed until later, but the sequence in which areas are targeted for protection can have a major effect on the cost-effectiveness of an overall conservation program.

Examples of targeted landscape management

Complex, multifaceted concepts like targeted landscape management are often more effectively illustrated through specific examples. Following are several examples that demonstrate new ideas, successes, and different approaches to incorporating targeted management into environmental protection.

Redefining conservation buffers in a landscape context

Water quality protection. Conservation buffers, such as filter strips and riparian forest buffers, can revitalize a host of physical conditions and ecological processes. Buffers can be especially important in extensively cropped landscapes where permanently vegetated areas are scarce. The effectiveness of conservation buffers depends upon the spatial and temporal juxtaposition and interactions between buffer zones and adjacent agricultural areas. For control of nonpoint-source pollution, buffers must be located where they will intercept agricultural runoff; trap a portion of its pollutant load; and stabilize, sequester, and transform those pollutants. General guidelines for federally funded conservation programs target the lower margins of source areas, such as cultivated fields and riparian zones along lake shores, wetlands, and streams. Buffers placed at field margins are closer to the runoff source. Depending upon site conditions, riparian areas can intercept and treat both surface and groundwater runoff.

This configuration of buffer strips woven through an agricultural landscape was recommended in the mid-1990s as an alternative to block configurations for CRP enrollment. But there are finer scale spatial patterns in agricultural landscapes that largely determine the water quality impact of buffer installations. Sources, transport pathways, and buffering suitabilities vary across landscapes, reflecting both inherent site properties and management influences. For example, source-area contributions vary with soil and slope properties of fields, as well as with tillage and other land treatments (e.g., Wischmeier and Smith, 1978). Overland flow is far from uniform, flowing into topographic swales, being diverted into drainage tiles and ditches, and even steered by the unevenness created by tillage furrows and ridges (Bren, 1998; Dosskey et al., 2003; Souchere et al., 1998; Tomer et al., 2003). Buffer effectiveness also varies with slope, soil type, groundwater depth, and subsurface geomorphology. In general, buffer capabilities may be greater in the upper reaches of watersheds than along the main stems of streams (Burkart et al., 2004). Buffer interactions with sources and pathways also occur, as pollutant-trapping efficiency largely varies with the amount and timing of intercepted runoff (Dosskey et al., 2002). Consequently, opportunities to effect watershed water quality are greater at some locations in a watershed than others. Targeting buffers to locations where impacts are likely to be greatest and avoiding those where impacts are likely to be small promises to improve substantially the conservation efficiency of buffers at the watershed or landscape scale.

Methods have been developed for assessing landscape patterns and identifying critical buffer locations. Ranking source areas by analysis of inherent site conditions and management, for example, by using USLE- or CREAMS-based models, has long been a focus of water quality improvement projects. Recent approaches have been developed that identify overland flow pathways and buffer suitability patterns. Terrain analysis using digital elevation models can identify probable patterns of runoff pathways and, at the stream-reach scale, relative baseflow contributions to streams (Tomer et al., 2003; Burkart et al., 2004). Topographic analysis also helps predict where variable-source areas, which generate overland flow nearest the stream, will occur (e.g., Lyon et al., 2004). Upslope contributing area and local slope of riparian areas are the most critical topographic parameters for making these determinations. Soil surveys can be used to identify the best-suited locations for buffer filtering of surface runoff using a model that accounts for both potential source size and buffer capacity (Dosskey et al., 2006). Soil surveys can also be used to identify locations where shallow groundwater occurs, which may be treated by buffers (Dosskey et al., 2006; Gold et al., 2001; Rosenblatt et al., 2001). Regional geomorphic patterns have been interpreted to assess the potential for riparian buffers to impact runoff and baseflow water quality (Lowrance et al., 1997). Landscape-scale geomorphic patterns may also be interpreted for potential nitrate attenuation in riparian groundwater (Vidon and Hill, 2006). At an even finer scale, a precision approach to buffer site design has been proposed that combines a spatial assessment of source-area loads, runoff pathways, and buffering capacities to recommend appropriate and varying buffer widths across landscapes (Dosskey et al., 2005). These methods all aim to identify locations in a landscape where buffer impact on water quality is likely to be greater than in others.

A buffer in an agricultural landscape for non-point-source pollution control that is based on recent science is likely to look somewhat different than the buffer strip model of the 1990s (Figure 1). Buffers should be most consistently located along the upper reaches of the watershed (Burkart et al., 2004). Wider buffers should be installed where surface runoff loads will be greatest or variable-source-area hydrology is most active (e.g., Walter et al., 2005; Qui, 2003). Riparian buffers need not be continuous or of constant width, but concentrated in areas where surface runoff is most readily intercepted and filtered, where saturation overland flow is likely to be generated from variable-source areas, and where buffer vegetation may have the greatest opportunity to influence shallow groundwater. There may be opportunities to combine topographic analyses and soil survey information in a complementary fashion to further enhance targeting of conservation buffers (Tomer et al., 2006).

The gains in conservation efficiency expected by targeting water quality buffers will be moderated somewhat by issues that include stratigraphic and pedogenic controls on subsurface flow patterns (Kung, 1990; Lowrance et al., 1997; Simpkins et al., 2002; Agier et al., 2005) and effects of hydrologic modifications, particularly artificial drainage (Wu and Babcock, 1999; Goehring et al., 2001; Dosskey et al., 2003). One solution to address artificial drainage is to incorporate constructed wetlands into riparian buffers that intercept and treat subsurface drainage water before it reaches streams (Schultz et al., 1995). Alterations of topographic and hydrologic patterns in intensively cultivated landscapes will probably complicate the use of existing assessment methods. For targeting techniques to overcome these problems, detailed and recent landscape data may be required for field-scale planning.

Habitat. Conservation buffers can create habitat that bolsters wildlife populations in agricultural landscapes. There is often too little suitable habitat, especially where intensive agricultural activities and infrastructure occupy most of the landscape. Remnants of native habitat may be too small to support viable wildlife populations. Isolation of habitat fragments restricts or prevents daily movements between sources of food, water, and cover; annual migrations to reproduction areas; and movement from remnants to newly

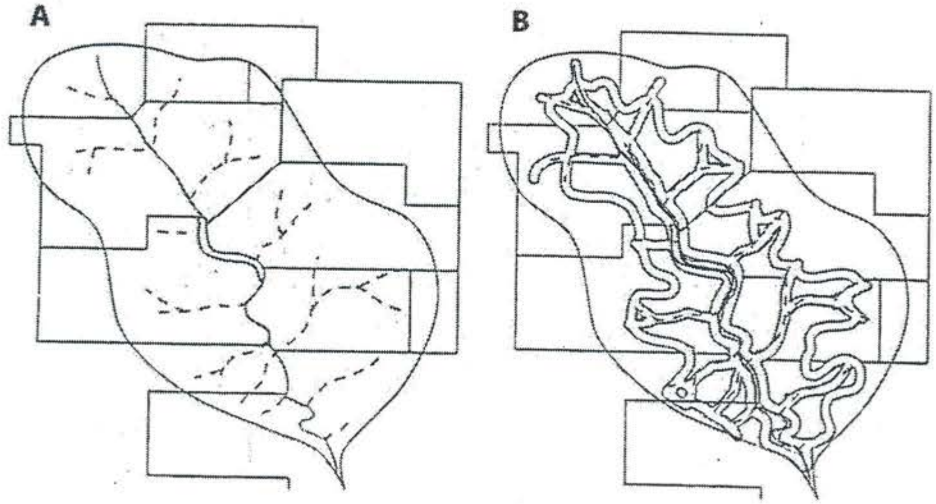


Figure 1. A conceptual diagram comparing (A) a traditional CRP block enrollment pattern in a watershed with (B) an area of land similar to a block, but set aside in a configuration of buffer strips (as shown in National Research Council, 1993). This diagram illustrates a recommended shift in approach to land retirement from blocks to targeted strips in order to increase the conservation effectiveness of CRP and related programs.

planted habitat areas. Buffers can help bolster viable wildlife populations by increasing the total area of suitable habitat, reducing the isolation of habitat remnants, and providing a management focus for improving habitat quality.

Greater wildlife impacts can be obtained from conservation buffers by appropriately locating them within agricultural landscapes. Corridors are a particularly important configuration in fragmented landscapes for increasing the viability of small, isolated patches; providing access to food, water, cover and other critical needs that may exist in different parts of the landscape; and creating avenues for dispersal and repopulating new habitat areas (U.S. Department of Agriculture-Natural Resources Conservation Service, 1999). Riparian areas offer disproportionately higher overall habitat value than upland areas (Naiman et al., 1993), in part because of their intrinsically greater habitat quality and their continuity through landscapes. Using conservation buffers to close existing gaps between habitat patches and create continuous riparian corridors can enhance conservation efficiency for terrestrial wildlife. GIS, coupled with land use and land cover maps, is a particularly helpful tool for determining critical locations (e.g., Bentrup and Kellerman, 2004)

Riparian areas are also major habitat elements for fish assemblages in streams and rivers (Schlosser, 1991). Adjacent vegetation affects channel form and structural diversity by contributing large organic debris to channels and by its influence on bank erosion and sediment deposition patterns (Trimble, 2004). Substantial inputs of organic matter and nutrients from riparian zones fuel the aquatic food chain (Vannote et al., 1980).

Opportunities for conservation buffers to benefit aquatic resources are generally greater in the upper reaches of stream systems, where interactions between riparian and aquatic systems are greatest (National Research Council, 2002). Shading effects on water temperature and accumulations of large woody debris are greater in smaller streams (Harmon et al., 1986; Hewlett and Fortson, 1982; Karr and Schlosser, 1978). Smaller streams also derive a greater proportion of their fish food resources from riparian areas (Newbold et al., 1980) and comprise a much greater collective length of streams in watersheds.

The conservation efficiency of buffers can be enhanced further by integrating placement criteria to benefit multiple resources. Riparian areas and corridors that connect them to remnant upland patches are particularly valuable for creating terrestrial habitat. Riparian areas in the upper reaches of watersheds are particularly influential on aquatic habitat and nonpoint-source pollution. Finer scale targeting, using topographic and soils information, should enhance the effectiveness of conservation buffers for improving water quality.

Hydrologically sensitive areas

One of the many new perspectives to come out of the New York City watershed protection program (Walter and Walter, 1999) is the concept of hydrologically sensitive areas—those areas especially prone to generating runoff (New York City Department of Environmental Protection, 1991; Walter et al., 2000, 2001; Agnew et al., 2006). The hydrologically-sensitive-areas management concept is simply to avoid potentially polluting activities (e.g., manure spreading) on areas most prone to generating runoff or to target water quality protection strategies on those areas that are both hydrologically sensitive and potential nutrient- or pollutant-loading areas. Some hydrologocially sensitive areas, such as paved barnyards, are obvious. Throughout most of the northeastern United States, however, runoff is generated from areas that become saturated. These areas can be very dynamic, expanding during wet periods and shrinking or disappearing entirely during dry periods; these dynamic areas are referred to as variable-source areas (e.g., Hewlett and Hibbert, 1967; Dunne and Black, 1970a,b) (Figure 2a). Research efforts as part of the New York City watershed program demonstrated that distributed models could accurately locate variable-source areas (Frankenberger et al., 1999; Mehta et al., 2004; Gerard-Merchant et al., 2006) and that by adopting associated targeted BMPs, phosphorus loading to streams could be reduced (Walter et al., 2001; Bishop et al., 2005; Hively et al., 2006). However, linking the basic hydrologic science describing where and when these areas will appear with policy and management structures has proven the most vexing challenge (Gburek et al., 2002), in part because hydrologically sensitive areas are both spatially and temporally distributed (Figure 2b).

Initial attempts to develop hydrologically-sensitive-area guidelines were incorporated into the New York phosphorus index (Geohring et al., 2002; Czymmek et al., 2003) by using a combination of soils information (e.g., flooding frequency) and set-back distances from identifiable runoff flowpaths (e.g., ephemeral streams). Even these initial achievements were a marked change from the original runoff-risk criteria

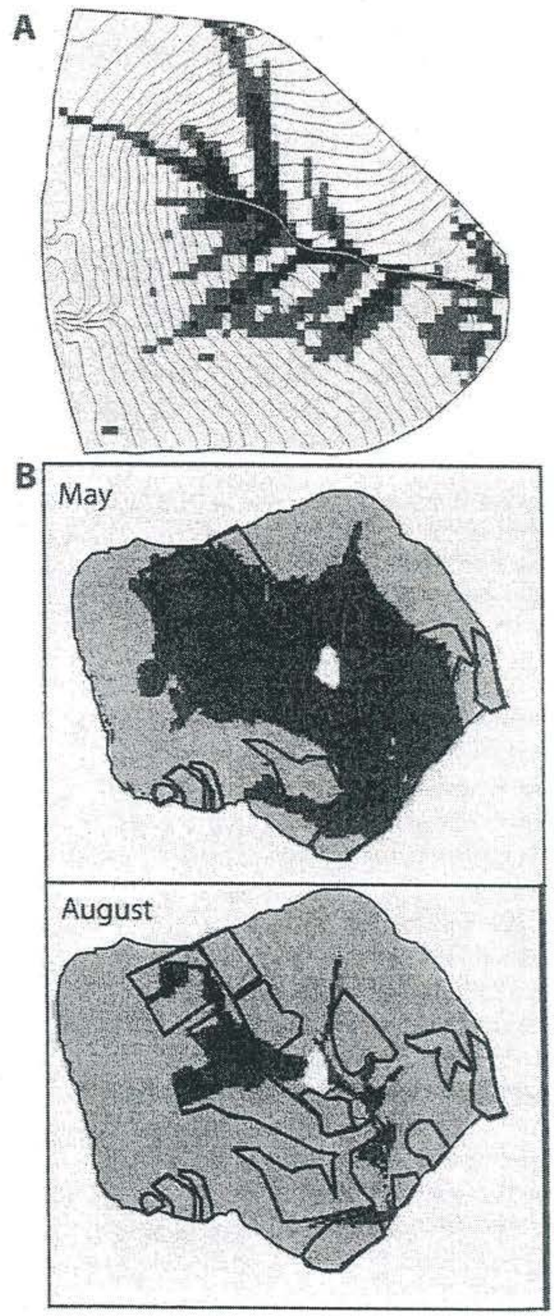


Figure 2. Examples of how variable source (runoff generating) areas (A) are distributed across the landscape (from Tomer, 2004) and (B) vary over the course of a year (from Agnew et al., 2006). In A, dark-gray and medium-gray areas represent frequently and occasionally runoff generating areas, respectively. In B, darker gray areas represent areas generating runoff at least five days in a month and the outlined areas delineate fields.

proposed for phosphorus indices, which correlated steep slopes and low infiltration capacities with high runoff potential (U.S. Department of Agriculture-Natural Resources Conservation Service, 1994). Neither of these factors correlates well with field observations of runoff-generating areas in the northeastern United States (e.g., Dunne and Black, 1970a,b; Frankenberger et al., 1999; Walter et al., 2003; Mehta et al., 2004; Easton et al., 2006). Interestingly, 84 percent of the current phosphorus indices have adopted hydrologically-sensitive-area-type criteria for identifying areas of high runoff risk (Gburek et al., 2006). One challenge in defining an hydrologically sensitive area is specifying a risk level, although researchers have demonstrated that levels can be quantified by distributed modeling (e.g., Walter et al., 2000, 2001), field measurements (e.g., Lyon et al., 2006a,c), and remote sensing (e.g., Verhoest et al., 1998). Several efforts are underway to identify and quantify hydrologically sensitive areas using GIS (e.g., Agnew et al., 2006), including internet-based tools, such as Google Earth (Lyon et al., 2006b). Additionally, local agencies (e.g., Tompkins County, New York Department of Environmental Planning) have begun requesting GIS data for their counties, in part as a means for delineating riparian buffers (Walter et al., 2005). These tools provide planners with information about how hydrologically-sensitive-area risk is distributed across the landscape and throughout the year. Consistency of data in terms of source and quality will remain a challenge to devising targeting tools based on hydrologically-sensitive-area assessments that can be consistently applied from a policy standpoint. Such tools will require a certain level of professional judgment in implementing targeted management in hydrologically sensitive areas.

Targeting criteria based on soil-test phosphorus

High livestock densities and associated land-applied manure pose serious threats to water quality across the United States and Canada (Sharpley et al., 1998; Canada-Alberta Environmentally Sustainable Agriculture Agreement Water Quality Committee, 1998). Livestock manure is typically applied to cropland based on crop nitrogen requirements. However, because the nitrogen-to-phosphorus ratio in manure is substantially lower than that of crops, accumulation of phosphorus on manured cropland is common (Sharpley et al., 1998). Surface runoff from cropland high in soil phosphorus can be a primary source of phosphorus enrichment of surface waters and subsequent eutrophication risks.

In Alberta, Canada, planners are developing targeted manure management that limits application based on site-specific soil-test phosphorus levels (Jedrych et al., 2006). Targeting was at the soil polygon scale because this scale represents the most detailed level of available soils information. Allowable soil-test phosphorus levels (modified Kelowna method) were quantified using the Water Erosion Prediction Project (WEPP) model to predict surface runoff, assuming water quality objectives of 0.5 milligram per liter and 1.0 milligram per liter and employing empirical relationships between total phosphorus concentrations in runoff and soil-test phosphorus in the top 15 centimeters (6 inches) of soil (Little et al., 2006). This method allows higher soil-test phosphorus values and higher associated total phosphorus in runoff for fields with lower runoff potential, and visa versa for fields with higher runoff potential. The spatial distribution of soil-test phosphorus was markedly different between the strict and more relaxed water quality criteria, 0.5 milligram per liter and 1.0 milligram per liter, respectively (Jedrych et al., 2006) (Figure 3). Perhaps most notably, soil-test phosphorus limits of less than 60 milligrams per kilogram (60 parts per million) were required for more than 80 percent of the agricultural land base in the province to meet the 0.5 milligram per liter water quality objective and dropped to less than 50 percent for the 1.0 milligram per liter objective (Figure 3).

The main advantage of this soil-test phosphorus targeting approach is that it uses the relative runoff potential, in this case determined by WEPP, instead of actual runoff values to calculate soil-test phosphorus limits; this eliminates the need for expensive WEPP model calibration. Unfortunately, because runoff estimates may be inaccurate, allowable soil-test phosphorus limits are similarly inaccurate. This approach, therefore, provides only a relative ranking of targeted priority areas.

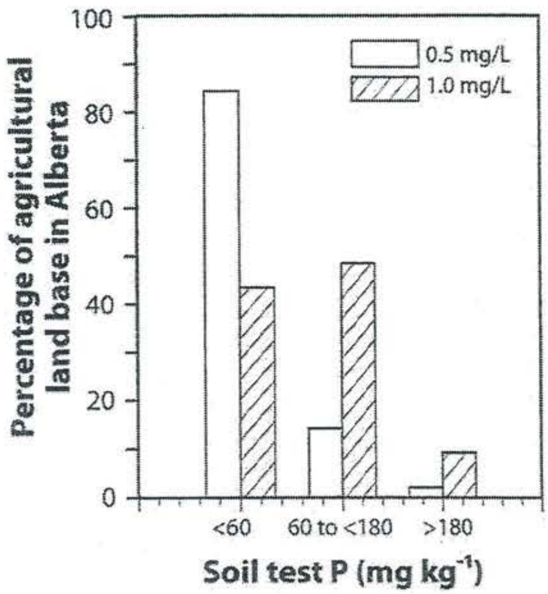


Figure 3. Soil test phosphorus limits for agricultural land base in Alberta required to meet total phosphorus water quality objectives of 0.5 and 1.0 milligram per liter.

Targeting places for biodiversity conservation

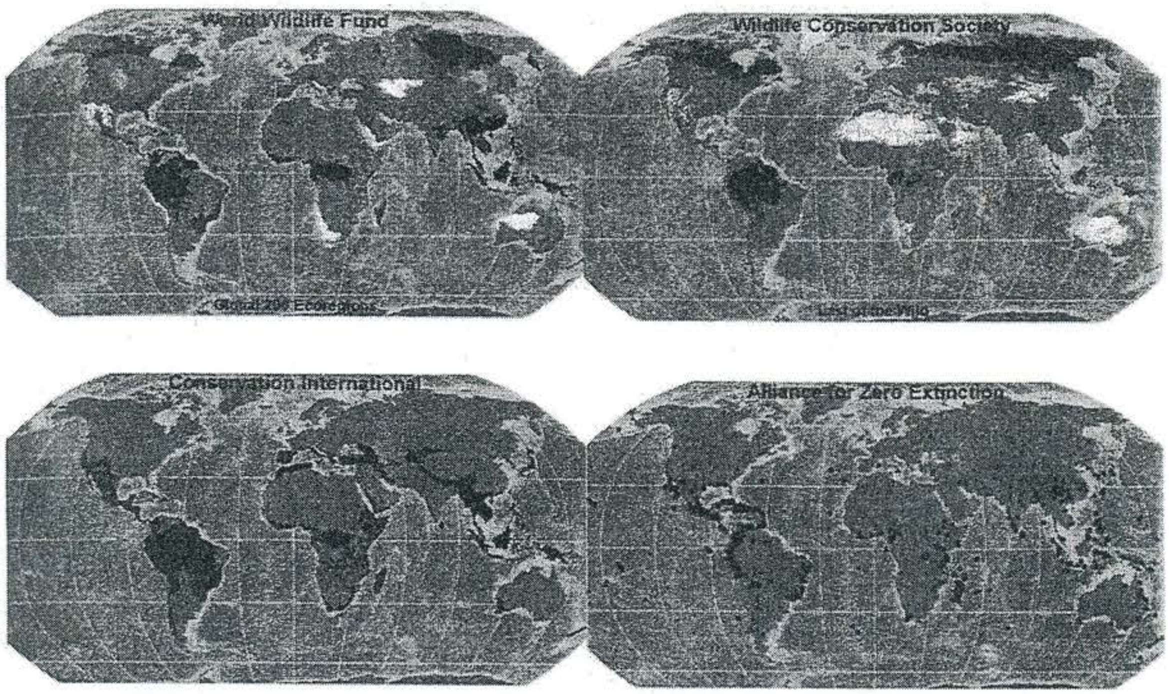
Recognizing that the needs for managing or protecting places for biodiversity conservation far outstrip the resources available to achieve that management or protection, several conservation organizations have used various approaches to prioritize (i.e., target) areas for concerted action, at scales from local to global. Rather than reviewing targeting approaches at the local or regional scales discussed in the previous examples, we briefly consider some efforts focused at a global scale. Groves (2003) provided an excellent review of targeting land management for conservation (a.k.a., conservation planning) at local and regional scales.

At a global scale, Conservation International has identified a number of "hotspots" of biodiversity that, if adequately protected, would contribute to preserving a substantial portion of the earth's species (Myers et al., 2000; Mittermeier et al., 1998). These areas are targeted largely on the basis of plant species richness (and secondarily vertebrates) and vulnerability. Not surprisingly, nearly all of the hotspots are in tropical and subtropical regions (Figure 4).

The World Wildlife Fund (WWF) has used species occurrences, the presence of distinct ecosystems and ecological processes, and the biological distinctiveness of areas to target a more widely distributed set of areas, the "Global 200 Ecoregions," for conservation action (Figure 4) (Olson and Dinerstein, 1998).

To address conservation priorities globally, TNC has developed an even more comprehensive, data-driven approach to targeting ecoregions within major habitat types of the earth (e.g., temperate conifer forests, deserts, and xeric shrublands), partitioned among biogeographic realms (e.g., Nearctic, Australasia). The approach emphasizes those ecoregions in which conservation actions may make the greatest contribution to the benchmark goal of effective conservation or management of 10 percent of a habitat type within a realm. Like the WWF Global 200, this approach is aimed at ensuring adequate conservation of representative biodiversity everywhere on earth – the "coldspots" as well as the hotspots (Kareiva and Marvier, 2003). Still other organizations have used different criteria to target areas for conservation at a global scale (Figure 4). Collectively, some areas are targeted by everyone, but others emerge only when representation is emphasized in the targeting process (e.g., Brooks et al., 2006).

All of these global targeting approaches emphasize in one way or another the spatial distribution of biological diversity and threats to that diversity at very broad spatial scales. TNC has taken an important step toward broadening the targeting criteria by using country-level data on political governance, economic risks, social well-being, and civil society development to define "enabling conditions." Not all places in which conservation is a high priority on biological grounds are conducive to investing in conservation or conducting work on the ground for sociopolitical or economic reasons. Moreover, conservation in most parts of the world cannot be achieved through strict protection of parks and reserves (although this certainly helps). Conservation must also be done in places where people live and work – people and their well-being must be part of the conservation equation and thus part of targeting land management (Miller and Hobbs, 2002; Millennium Ecosystem Assessment, 2005). One way to include social, political, and economic factors in conservation targeting is by using return-on-investment tools



Source: Maps courtesy of the Nature Conservancy.

Figure 4. Areas targeted for biodiversity protection by four conservation organizations using different global targeting criteria.

and models from economics to prioritize areas for conservation or the actions that might be taken to achieve conservation most economically in those areas. While this work is still in an exploratory phase (e.g., Wilson et al., 2006; Polasky et al., 2001), it already has shown that traditional (even sophisticated) ways of targeting places based on their apparent biodiversity value alone may not lead to the most effective use of limited funds. Combined with new methods of evaluating the success of conservation actions (e.g., Ferraro and Pattanayak, 2006), it may be possible to target conservation to achieve the greatest "biobang for the buck." Such prioritizations must include socioeconomic and political factors as well as the biodiversity value of targeted places.

Challenges and future directions

We mentioned previously the substantial progress that has been made toward developing rigorous and comprehensive conservation planning (targeting) at multiple scales. Yet the greater understanding of habitats and habitat relationships of organisms across those scales has also revealed several bothersome realities that raise doubts about how much we really do understand ecological systems. These realities, in turn, affect our ability to target management actions in ways that will actually produce desired outcomes. We note here two of these realities.

First, the elements of ecological systems rarely act in isolation, and management activities rarely have only a single consequence. There are interactions and interdependencies to consider. Sometimes these are complementary, for example, when habitat protection for one endangered species results in habitat protection for a suite of associated species, or when measures to enhance seasonal streamflow for fish migration also result in a flushing of sediment from basins and improved water quality and biotic integrity. In other situations, however, land management targeted by one set of criteria may have unintended consequences or negative impacts on other components of an ecological system. For example, use of prescribed burning to maintain an open understory in woodlands may favor hawks that prey on ground-dwelling mammals, but negatively affect not only mammal populations, but also bird species that require an intermediate shrub layer for nesting. Preventing human intrusion into a forested watershed that serves as a municipal water supply may protect water quality, but it may also preclude human recreation in the watershed, leading to reduced support for watershed conservation efforts. Conservation is increasingly about tradeoffs among competing interests or targets. Return-on-investment or cost-benefit analyses may provide pathways to quantifying or even optimizing these tradeoffs, but this requires knowledge of what to include in the analysis, how to weight the competing interests, and what metrics to use as measures of "return" and "investment." In the world of conservation, these metrics are often noneconomic, but the science of incorporating noneconomic factors into what are essentially economic models is not yet well developed.

The second reality has to do with "thresholds"–"tipping points" in contemporary parlance (Gladwell, 2002). Targeted land management is often conducted as if the targeted systems and the outcomes of targeted actions were stable, or at least varying within well-defined and prescribed limits. But ecological systems are dynamic in time and space. Not only that, they are inconsistently dynamic. Natural or human-induced changes in system properties that appear to be gradual and continuous may sometimes lead to sudden and irreversible shifts to some other system or set of prevailing conditions, as envisioned in so-called "state-and-transition" models (e.g., Bestelmeyer et al., 2003). Examples of such changes in arid lands, as from grasslands to shrublands or vice versa, are legion (e.g., Bestelmeyer, 2006), but they occur in virtually every type of terrestrial and aquatic habitat (e.g., Groffman et al., 2006). Sudden, irreversible changes resulting from global climate change are of increasing concern (e.g., Lovejoy and Hannah, 2005). The challenge to targeted land management is clear: Not only must we recognize the appropriate targets for our management actions to have the intended effects, but we are managing with reference to a moving target and (to stretch the metaphor) one that may change shape, identity, and context suddenly and without warning. A better predictive science of ecological thresholds would enhance targeted land management.

Summary and conclusions

In conclusion, we emphasize four points. First, although conservation practices have always had some degree of "targeting," targeting approaches have become increasingly refined with our improved understanding of how individual parts of the landscape are interconnected. Concurrently, technological advances in GIS, remote sensing, and other methods of gathering, compiling, and analyzing geospatial data have facilitated the linkage between science and the application of targeted land and water management for conservation. But there is a catch. Although the scientific bases for targeted protection are generally good and getting better, the ease with which GIS and other technologies can be used can facilitate the inappropriate use of older data and models. These may have been effective in meeting the objectives of their day, but they do not incorporate recent scientific advances that have created, for example, data with a high degree of spatial resolution at multiple scales or analytical procedures that can incorporate the nonlinear or threshold dynamics of ecological systems. As ever-more-powerful technological and analytical tools become available, it is important to keep in mind that such tools are only as good as the information they have to work with. Assumptions about the utility of data must be clearly stated. Great tools cannot produce great targeting with inadequate data, or with hidden assumptions about oversold data.

Second, determining how and what to target, with what degree of precision, should be driven by one's objectives. The targeting approach that one takes to evaluate the consequences of agricultural practices on stream water quality in the Midwest will likely differ from that applied to a similar situation in the Intermountain West, or in Guatemala or Bangladesh. The tools used to target an area for the application of soil erosion mitigation will likely differ from those used to target wildlife habitat conservation in the same area. There is no great insight here. But what this does highlight is the critical importance of clearly stating the goals and objectives of a conservation project, in operational terms, at the outset of any targeting effort. And the objectives must align with the characteristics of the biophysical system and the management strategies to be applied to that system (Figure 5).

Third, issues of scale are of overriding importance. The challenges to water quality conservation are not the same at the scale of a local farm, in which the primary concern may be with surface and subsurface runoff from manure spreading on fields into a small stream, as at the scale of a large regional watershed (e.g., the Upper Mississippi River), where a host of point and nonpoint sources act to influence water quality. Scientific work over the past two decades has shown the many ways in which both ecological patterns and processes and our perceptions of these patterns and processes are influenced by the scale of reference (e.g., Wiens, 1989; Peterson and Parker, 1998), although this work has not yet produced a workable way of predicting these effects. What is clear, however, is that targeting efforts must be scale-sensitive. This means that targeting must focus at scales where the scales of the system, management practices, and objectives coincide (Figure 5). Indeed, a goal of targeting should be to move these components into greater scale concordance (i.e., greater overlap in the Venn diagram of Figure 5).

Finally, our emphasis in this chapter has been on the science that underlies efforts to target conservation efforts to enhance the effectiveness (and cost-effectiveness) of management, mitigation, and restoration. But science alone does not have all the answers. Science can inform one about where conservation efforts might do the most good, when they would best be applied, and how to gauge whether or not they are working as desired. But whether or not the targeting recommended by science can or should be implemented involves other considerations. Land and water management are conducted in an economic, social, and political context, and this context is what determines whether the science-based course of action is in fact feasible or desirable. The mitigation or restoration of riparian areas as buffers against runoff from agricultural land, for example, may be too expensive for the benefits to be derived, or the lands identified through a habitat targeting procedure for wildlife protection may be unavailable. And we have already mentioned the bothersome issue of tradeoffs – a conservation action that is highly desired by some segments of society may be vigorously opposed by other segments. If compromises are possible, one must then consider whether the diminished

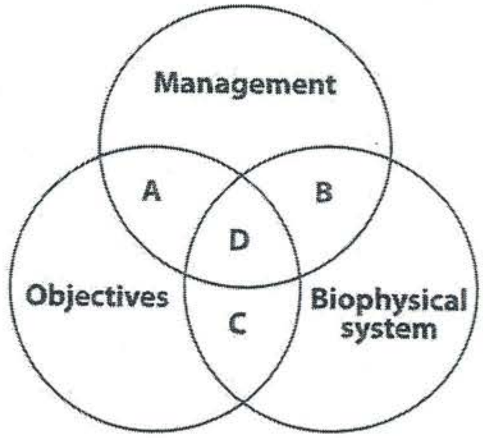


Figure 5. The intersection of objectives with management actions and the biophysical system under consideration. Targeting can aim to align objectives with management actions (A), management with the systems (B), or objectives with the biophysical system (C). Ideally, targeting should aim to align all three components simultaneously (D). Because objectives, management, and the biophysical system may be expressed or operate at different scales, targeting should also aim to align all three components within a common scale of reference (D).

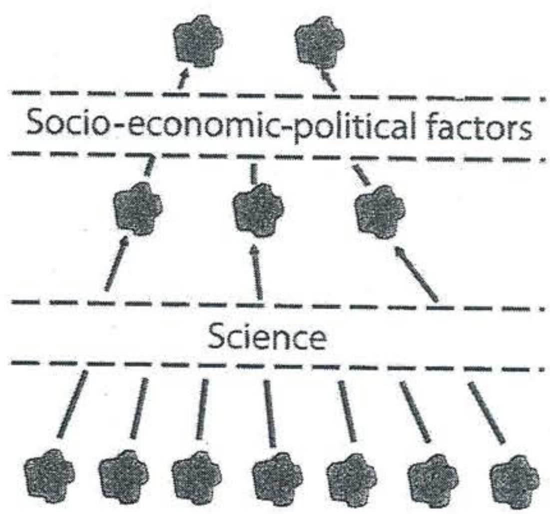


Figure 6. The role of science and socio-economic-political factors in targeting places for conservation. A set of possible locations for conservation action (bottom) is filtered by science-based targeting to identify a subset that will most effectively yield conservation benefits; socio-economic-political considerations then determine in which elements of this subset conservation efforts may be feasible or acceptable.

benefits make the actions worth the effort. These realities should not be taken as weakening the importance of science in targeting, however. In our view, science provides the foundation, the first-pass assessment of what should be done, based on a rigorous analysis of data from multiple sources. Once that judgment has been made, then the socioeconomic and political realities should be incorporated to determine which of the possible targeting actions identified by scientific analysis actually merits implementation (Figure 6).

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Roundtable: The science of targeting to improve conservation effectiveness

Frustration exists among many natural resource professionals over the difficulty of moving forward with targeting conservation in watersheds and on landscapes. Successful targeting in watersheds and on landscapes depends upon both biophysical and social factors:

- Biophysical information that identifies sensitive areas within landscapes.
- Behavioral information that explains why inappropriate land management occurs in sensitive areas and identifies factors that determine the willingness of land managers to adopt conservation in those locations.
- Program structure that encourages conservation adoption by landowners and managers in targeted areas.
- Policy that motivates professionals to target conservation to owners and managers of sensitive areas where greater environmental impact can be achieved.

Biophysical models appear well advanced and continue to refine our ability to identify sensitive areas within landscapes and watersheds. Existing models are underutilized, and emerging approaches and technologies promise to enable even greater conservation efficiency. But the power of biophysical models will be widely applied only if the knowledge contained in complex, research-type models is translated into simplified models that are easily used by natural resource professionals. This translation must be accompanied by assessments of uncertainty and limitations in the models and guidance on selection and use of appropriate spatial data from a rapidly growing and complex body of available sources.

Greater challenges involve the human dimensions of targeting where difficulties stem from behaviors of both land managers and natural resource professionals. In many cases, great strides in conservation effectiveness could be made by changing the management behaviors of a small fraction of land managers within watersheds or landscapes. Motivating these few land managers to adopt conservation practices by way of education and financial incentives has not been very successful, which suggests that there are other important factors that determine the willingness or unwillingness of key land managers to adopt conservation practices. Better understanding of the broader suite of motivational factors would guide improvements in incentive strategies. Effective strategies must encourage land managers to choose practices that accrue conservation benefits mainly to society (or to the watershed) over those that provide benefits mainly to the individual farm or ranch.

Policy and programs need to be structured in ways that encourage targeted application of conservation practices at the appropriate scale. Today, targeting often amounts to random application of conservation within large-scale problem areas, such as watersheds with total maximum daily load (TMDL) concerns, whereas finer-scale targeting may be necessary to achieve meaningful conservation impacts. Important strategies remain underutilized, such as flexibility in tailoring enrollment criteria, like the environmental benefits index, to meet local needs, and marketing conservation practices to managers in targeted areas while maintaining equal-access requirements. Greater use of existing targeting strategies could be achieved if there were policies that rewarded targeting efforts among natural resource professionals.

Finally, greater local control of targeting decisions may further improve conservation success. Local knowledge of sensitive areas and management behaviors can provide critical information at finer scales of resolution than generalized biophysical and behavioral models.

Top-down decision-making based on generalized models and enrollment criteria may be too coarse to identify key locations and motivate those land managers who can produce the greatest conservation impact.