Biomass and Bioenergy 128 (2019) 105327

Contents lists available at ScienceDirect

Biomass and Bioenergy

journal homepage: www.elsevier.com/locate/biombioe

Review

Designing bioenergy landscapes to protect water quality

Jasmine A.F. Kreig[a,b,*], Herbert Ssegane[c], Indrajeet Chaubey[d], Maria C. Negri[e], Henriette I. Jager[b]

[a] Bredesen Center for Interdisciplinary Research and Graduate Education, University of Tennessee Knoxville, Knoxville, TN, 37996-3394, USA
[b] Oak Ridge National Laboratory, Oak Ridge, TN, 37831-6038, USA
[c] The Climate Corporation, Monsanto Company, St. Louis, MO, 63167, USA
[d] Purdue University, West Lafayette, IN, 47907, USA
[e] Argonne National Laboratory, Lemont, IL, 60439, USA

ARTICLE INFO

Keywords:
Spatial optimization
Riparian buffers
Artificial drainage
Residue harvest
Biomass feedstocks
Scale

ABSTRACT

Practicing agriculture decreases downstream water quality when compared to non-agricultural lands. Agricultural watersheds that also grow perennial biofuel feedstocks can be designed to improve water quality compared to agricultural watersheds without perennials. The question then becomes which conservation practices should be employed and where in the landscape should they be situated to achieve water quality objectives when growing biofuel feedstocks. In this review, we focused on four types of spatial decisions in a bioenergy landscape: decisions about placement of vegetated strips, artificial drainage, wetlands, and residue removal. The appropriate tools for addressing spatial design questions are optimizations that seek to minimize losses of sediment and nutrients, reduce water temperature, and maximize farmer income. To accomplish these objectives through placing conservation practices, both field-scale and watershed-scale cost and benefits should be considered, as many biophysical processes are scale dependent. We developed decision trees that consider water quality objectives and landscape characteristics when determining the optimal locations of management practices. These decision trees summarize various rules for placing practices and can be used by farmers and others growing biofuels. Additionally, we examined interactions between conservation practices applied to bioenergy landscapes to highlight synergistic effects and to comprehensively address the question of conservation practice usage and placement. We found that combining conservation practices and accounting for their interactive effects can significantly improve water quality outcomes. Based on our review, we determine that by making spatial decisions on conservation practices, bioenergy landscapes can be designed to improve water quality and enhance other ecosystem services.

1. Introduction

The goal of sustainable management of multi-functional landscapes is to maximize future ecosystem services [1]. These include provision of water, energy, food, and feed, as well as regulation of water quality. Spatial decisions can play an important role in managing landscapes in a way that leads to increases in ecosystem services. Most problems in resource management are ultimately spatial optimization problems [2]. Land owners decide how to arrange managed lands on the landscape, where to place conservation elements, and how to manage different parcels. Among these decisions is the option to include biomass crops or residues. Spatial decisions that support biomass-feedstock production add bioenergy to the portfolio of ecosystem goods (e.g., food) and services (e.g., water purification) provided by the landscape.

Quite a few studies, many of which are reviewed here, have addressed the question of how to best manage land using decision tools, but very few provide general guidelines relevant to bioenergy producers, nor consider interactions among management practices. In many cases, studies have focused on the methods used to find solutions to optimization problems, rather than on the solutions themselves. Because solutions are often site-specific, it is challenging to extract generalizations that apply to other situations and locations. Decision support systems built on process-based models that require site-specific data are not feasible for all situations and scales. The goal of this paper is to extract generalizable rules of thumb from case studies, to begin to fill this void in the literature.

In this review, we synthesize optimization research to guide spatial decisions for making field- and watershed-scale management decisions leading toward improved water quality in landscapes that produce biofuel feedstocks. Papers that came up as a result of using search terms

* Corresponding author. Bredesen Center for Interdisciplinary Research and Graduate Education, University of Tennessee Knoxville, Knoxville, TN, 37996-3394, USA.
E-mail address: jkreig@vols.utk.edu (J.A.F. Kreig).

https://doi.org/10.1016/j.biombioe.2019.105327
Received 16 November 2018; Received in revised form 18 July 2019; Accepted 1 August 2019
Available online 07 August 2019
0961-9534/ © 2019 The Authors. Published by Elsevier Ltd. This is an open access article under the CC BY license
(http://creativecommons.org/licenses/BY/4.0/).

such as "bioenergy", "water quality", "optimization", "perennial grasses", "buffer strips", "tile drains", "wetlands", and "residue removal" were selected for this review. We classified studies by spatial scale, management practices, and two objectives (improve water quality and maximize farmer profit). Results were synthesized using decision trees to provide context-dependent guidelines that can be used when designing bioenergy-producing landscapes.

We focused on four types of spatial decisions that influence lignocellulosic biomass from switchgrass, miscanthus, native prairie grasses, short-rotation woody crops, and corn stover: (1) where to place vegetated filter strips (a.k.a. buffers), (2) where to place (and how to manage) artificial drainage systems, (3) where to restore or create wetlands, and (4) where and when to harvest residues. For each type of decision, we evaluate how these can maximize water quality as an objective, among other objectives, such as maximizing farmer profit. In this review, we focus on temperature, total suspended sediment (TSS), total nitrogen (TN), and total phosphorus (TP) as aspects of concern when improving water quality. This is because there are thresholds for these factors that, when exceeded, cause harm. We also discuss differences in configurations that emerge from different types of spatial decisions and the contextual variables that can lead to better decisions. We finish by discussing interactions among the four spatial decisions outlined above, as such decisions on land management are not made independent of landscape characteristics or other management practices. Such interactions between the four spatial decisions are also summarized in an interaction matrix (see Section 6).

We discuss decisions made at the watershed-scale (Fig. 1) and field-scale. These two spatial scales involve different decision makers and processes. Integrating bioenergy crops into current agricultural systems at field- and watershed-scales can increase nutrient use efficiency at the field-scale and contribute overall to better water quality outcomes [3,4]. Decision makers may assign different priorities to different land-management objectives. Multiple crops can exist on those landscapes, but here we focus on those landscapes include biomass feedstock production. At the watershed-scale, decisions that influence water quality involve a significantly larger number of stakeholders, likely with different priorities [5]. At the field scale, individual farmers and landowners make decisions that can be influenced by personal preferences, policies, and economic constraints [6–8]. The issue of profitability (i.e. how productive farmland is) also factors into decision-making, with farmers choosing to plant crops where they will bring the highest profit [6–9].

In this paper we assume there will be a bioenergy market in the future [10]. We focus here on two on two ecosystem services in this analysis: water quality improvement and biomass-crop harvested for ethanol production. Decisions at broader extents, those that set state or federal policies for example, can also influence local decisions about land-management that affect water quality.

2. Spatial decisions about vegetated filter strips

Integration of native perennial grasses into agricultural crop production systems can help to remove sediments, nutrients, and other pollutants from agricultural runoff by slowing flow velocity and thus increasing water infiltration [11]. In this paper, we will use the term 'filter strip. For example, contour buffer strips are grass filter strips placed along a contour alternating with wide bands of cropland, whereas riparian buffers are filter strips planted shrubs/trees along streams or other water bodies and often include bioenergy crops.

The performance of filter strips, defined as improved water quality, depends on where they are placed. Spatial context includes land use, soil type, and presence of tile drains, at both local field and watershed scales. In referring to optimization studies, it has been seen that on individual fields, spatial decisions (e.g., about landscape position) seek

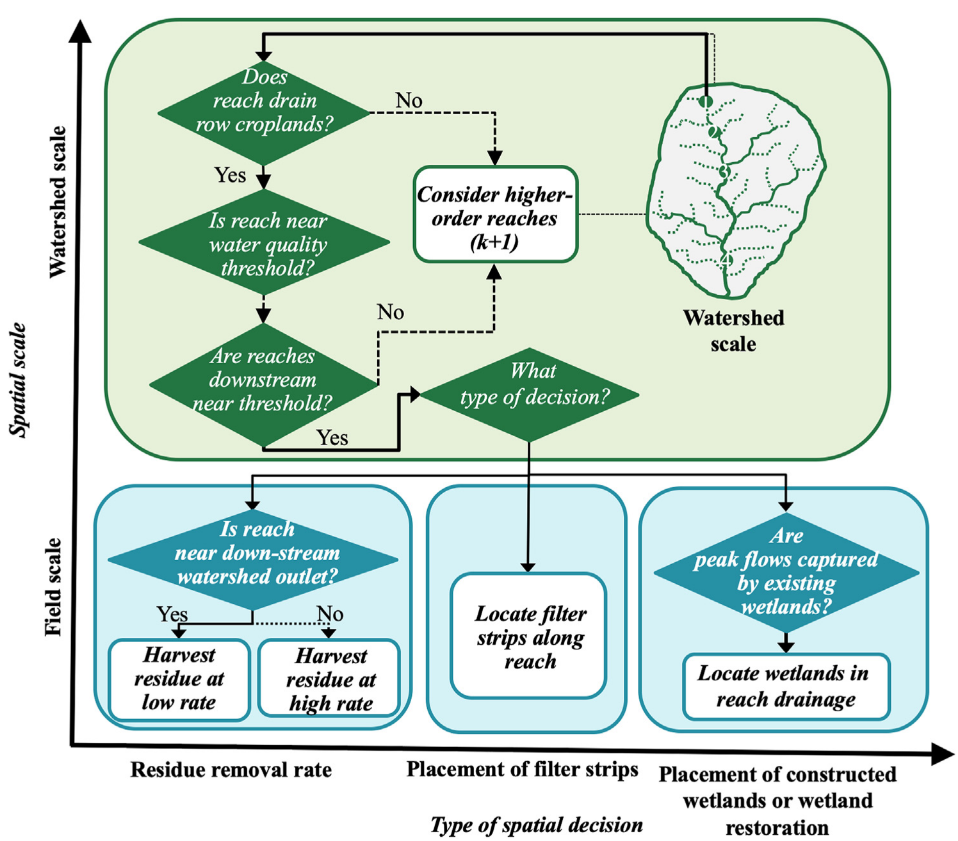


Figure 1. Decisions at the watershed scale that are examined in this review. Field scale decision trees exist for each type of spatial decision in their respective sections.

to minimize edge-of-field sediment and nutrient export (field-scale deployment) [12], whereas deployment at the watershed scale typically targets critical source areas [13]. Performance at each scale depends on the spatial design in juxtaposition to landscape features and properties (i.e., covariates).

2.1. Watershed-scale placement decisions

Spatial decisions about placement of filter strips and buffers in a watershed are relevant both in agricultural [3,11,14,15] and forested [16,17] production systems. These can help to moderate river temperature, minimize flooding, and reduce erosion and nutrient runoff, and if harvested, can contribute income to producers. Examples include harvested willow [3] and corn stover [9]. Below, we summarize conditions that favor upstream versus downstream position of buffers from optimization studies conducted at a watershed scale.

2.1.1. Conditions favoring upstream placement

Water-quality improvement measures that are implemented upstream propagate downstream so that overall benefits are magnified [14,18]. For example, when the objective is to lower water temperature, shading is more effective in smaller, headwater streams than in wider mainstems downstream [19]. Similarly, filter strips on upslope fields of tributary streams provide greater cumulative effect on improving water quality in a watershed than those located downstream (Fig. 2). Several optimization studies have concluded that placing filter strips and buffers in source areas of headwater watersheds is a good strategy [3,14]. Contour and vegetative strips in multiple headwaters were more effective than one downstream riparian strip because most runoff was captured by headwater streams before flowing to a larger river [14]. Concentrated flows can occur as runoff moves from upland plains to flood plains or rivers if there are no flood plains. This results in powerful erosive forces and could be mitigated by planting bioenergy buffers of switchgrass (*Panicum virgatum*) and willow (*Salix alba L.*) to intercept and shorten flow path lengths [3]. This is a great option for achieving concurrent environmental and biomass benefits because the buffers can be harvested for biomass feedstocks. The marginal cost of adding buffers in additional catchments suggested that targeted placement of buffers in watershed was a cost-effective strategy [19].

Spatial decisions at the watershed scale impact water quality at drainage outlets. Using a calibrated Soil and Water Assessment Tool (SWAT) model, Jha [20] projected reductions in nitrate loadings due to conversion of row crops to grasslands for three areas. Results showed larger reductions when grasslands replaced crops in highly erodible

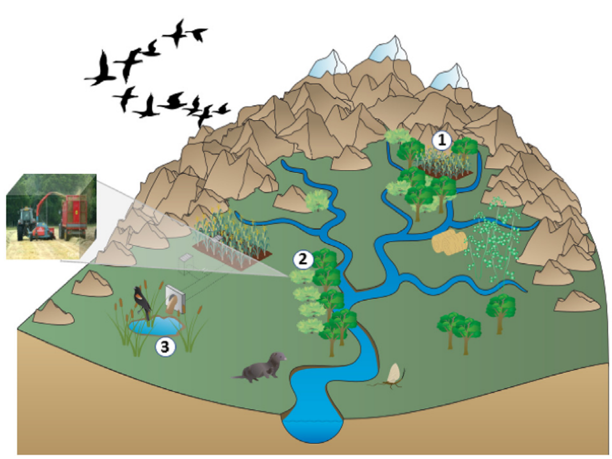


Fig. 2. Illustration of several principles of vegetated filter strip design include 1) placement in uplands and upstream tributaries, 2) multi-layer buffers with harvesting of outer layers, and 3) placement of mitigation (wetland and vegetation) at the outlet of tile-drains.

areas, or critical source areas 47% reduction, than when converting crops to grassland in upstream parts of the watershed (16%) or in downstream floodplains (8%) [20]. Additionally, Ha and Wu [21] reported that, compared with adding buffers in the mainstem, adding buffers in tributary streams caused a larger decrease in nitrate than in sediment or TP. This was true not only at the outlet, but also across the basin. Recently, the SWAT model was used to evaluate filter strips as conservation measures for two river basins under a projected 2040 bioenergy scenario [10]. In the Iowa River, simulations of 30-m and 50-m buffer strips in a corn-dominated agricultural setting were simulated [22]. The study estimated that applying a 50-m buffer to the entire stream network in the Iowa River Basin would increase coverage to 11.3% of the land and thereby decrease nitrate by 10.8% and sediment by 80% [22]. The study estimated that the feedstock production in the Iowa River basin from corn stover, willow, and miscanthus would total 121,000 metric dry ton [22]. Furthermore, if 50% of the switchgrass grown along 125 km of stream were harvested, an additional 121 million dry metric-tons of biomass could be obtained [22]. These results were similar to those found in a smaller subbasin, the South Fork Iowa River basin [21].

2.1.2. Conditions favoring downstream placement

Filter strips can be placed in flood-prone areas to prevent crop losses due to flood damage. When harvested for biomass, perennial filter strips may, in future, provide income to farmers [9]. As an example, Li [23] found that excessive rainfall and flood damage in the US has reduced corn yield by 34% over past decades. In other cases, unproductive areas within fields may further degrade by losing topsoil when planted in annual row crops. This is in contrast to perennial lignocellulosic biomass crops, which can be successfully grown on these lands [24]. For example, 150 ha of 6-m wide grassed buffer strips were planted along field edges in a watershed as part of an integrated approach to control flooding [25,26]. Buffer strips of biomass crops may help to prevent soil degradation and, in future, may supplement farmer income [19,21,27–29] once lignocellulosic feedstocks become cost-competitive.

The link between degraded water quality and the percentage of agricultural land is well know [30]. Because prime agricultural lands tend to occupy areas with lower slopes, the best opportunities for buffering stream networks may therefore present themselves downstream where agriculture is a dominant land use. In an optimization using the SWAT model, replacing more-intensively-managed crops (e.g., crops that require high levels of fertilizer, etc.) with switchgrass in downstream areas of an East Tennessee, US watershed was found to be an effective strategy for improving water quality [27]. Concentrated planting (less than 2% area) of perennial switchgrass near the outlet of an agricultural watershed decreased TN by 0.08 $mg\,L^{-1}$, TP by 0.02 $mg\,L^{-1}$, and TSS by 25 $mg\,L^{-1}$ compared to a baseline landscape [27]. In another modeling study [22], the largest reductions occurred in the middle of the Iowa River watershed where the highest acreages of annual crops occurred.

2.1.3. Summary

To summarize criteria related to siting filter strips at the watershed scale, different objectives may favor different solutions (Fig. 3). The risk of erosion is higher in uplands, whereas flooding risk is higher in lowlands, suggesting that placement may depend on which objective (flooding or erosion) is more important. Upstream placement along tributaries tended to be favored in optimizations focused on moderating stream temperatures or reducing sediment and nutrient losses. Downstream placement was favored when tributary basins were not in cultivation or when the objective was to minimize flooding.

2.2. Field-scale placement decisions

Spatial decisions regarding placement of filter strips within

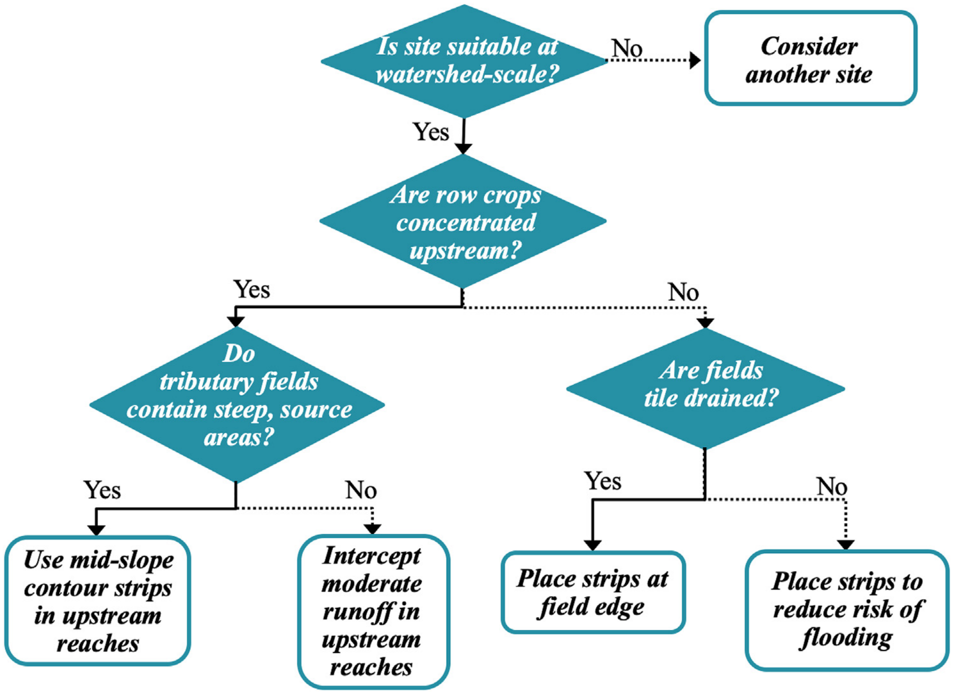


Fig. 3. Decision tree for siting filter strips when considering water quality objectives and field attributes.

individual agricultural fields are driven by terrain and hydrology (e.g., slope and surface flow pathways) and soil erodibility. Site selection can be aided by simple geospatial assessments with the help of modeling tools [11,31]. Although reducing losses of sediment and nutrients are primary objectives of using filter strips, other objectives, such as biomass harvest to augment profit to producers, may also play a role. Below, we discuss two other considerations that farmers use when deciding where to place filter strips: water quality and profit. Finally, we discuss the design of multi-tiered buffers that meet multiple objectives.

2.2.1. Decisions that maximize water quality

The effectiveness of filter strips to trap nutrients, sediment, and other contaminants depends on site specific factors such as filter-strip width, slope, vegetation, and the influent nutrient loading [32]. Most filter strip designs focus on using models to determine the optimal buffer width [33,34]. For example, a review by Fischer and Fischenich [33] recommended a 5-m to 30-m width for water-quality protection. However, the optimal width of filter strips may depend on context, as was the case in a field study on non-tile-drained lands. In this study, small conversions (10%) of agricultural fields to filter strips of widths 37.6 m–78.2 m of native prairie grasses planted along the contour and at the foot slopes reduced sediment, total nitrogen, and total phosphorus by 95%, 85%, and 90%, respectively [35].

Slope can greatly affect the effectiveness of filter strips in trapping nutrients at the field scale. For example, on sloped fields, filter strips planted along slope contours are effective in interrupting surface flow [36]. Sahu and Gu [37] used a calibrated SWAT model to illustrate that strips planted mid-slope on a contour were much more effective in reducing nitrate export than those planted downstream along the riverbank.

Finally, the type of load and location of influent should be considered when siting filter strips. According to Bentrup [38], the areas separating runoff-generating areas and pollution-loading areas are the most critical to buffer. Locating filter strips near waterbodies is often recommended, in part to avoid interference with farm operations, and in part to intercept flow prior to affecting downstream receiving waters. Cost-benefit assessments by Geza [28] showed that for a given amount of sediment reduction, planting filter strips along field drains (edge-of-field) was less expensive than planting them upslope in critical sediment source areas.

Generalizing recommendations such as those above requires that we develop appropriate relationships with spatial covariates. Commonly used models (e.g., the filter strip model, VFSMOD [39] and the Riparian Ecosystems Management Model (REMM) [40]) integrate factors, such as slope and erodibility that influence filter-strip performance. These depend on relatively simple relationships that have been derived between variable buffer width (W [m]), buffer slope (S [%]) and a soil erodibility (E). For example, studies reported $W = (30.5{*}S^{1/2})/E$ [41] and $W = 8 + 0.6{*}S$ [42]. Ssegane et al. [3] assessed filter strip performance in Illinois, which was demonstrated to be a function of strip width and trapping efficiencies (Fig. 4). Wider filter strips (6–10 m) produced higher trapping efficiencies than narrower strips (2 to 5-m) for runoff but not for sediment trapping. However, neither runoff interception nor sediment trapping differed between the 6–10 m and wider buffers. For nitrate and pesticides, buffer trapping efficiencies did not increase significantly for widths greater than 5 m. These results suggest that widths of 6–20 m are adequate to meet over 70% reduction in nutrient and pesticide transport (Fig. 4). On average, a switchgrass or willow buffer could reduce annual leached NO_3 by 61 or 59% and N_2O emission by 5.5 or 10.8%, respectively [3].

2.2.2. Economic decisions

The economic impact of filter strip placement is an important consideration. Decisions that influence economic outcomes involve 1) opportunity cost of planting perennials where row crops can be profitably planted, 2) income from harvesting biomass along filter strips, 3) risk of flood damage to crops that are intolerant of wet soils, 4) inefficient equipment paths caused by filter strip placement, 5) interference of filter strips with drainage systems, 6) effort and cost of maintaining filter strips compared to managing row crops, and 7)loss of productive land to filter strips. A full techno-economic analysis for different placement decisions is not possible at this point, but based on our review, we address a few of these decisions below.

Farmer decisions about filter strips depend on whether they can be placed where non-bioenergy crops are unprofitable and whether the perennials used in strips can be harvested for profit. These concerns are

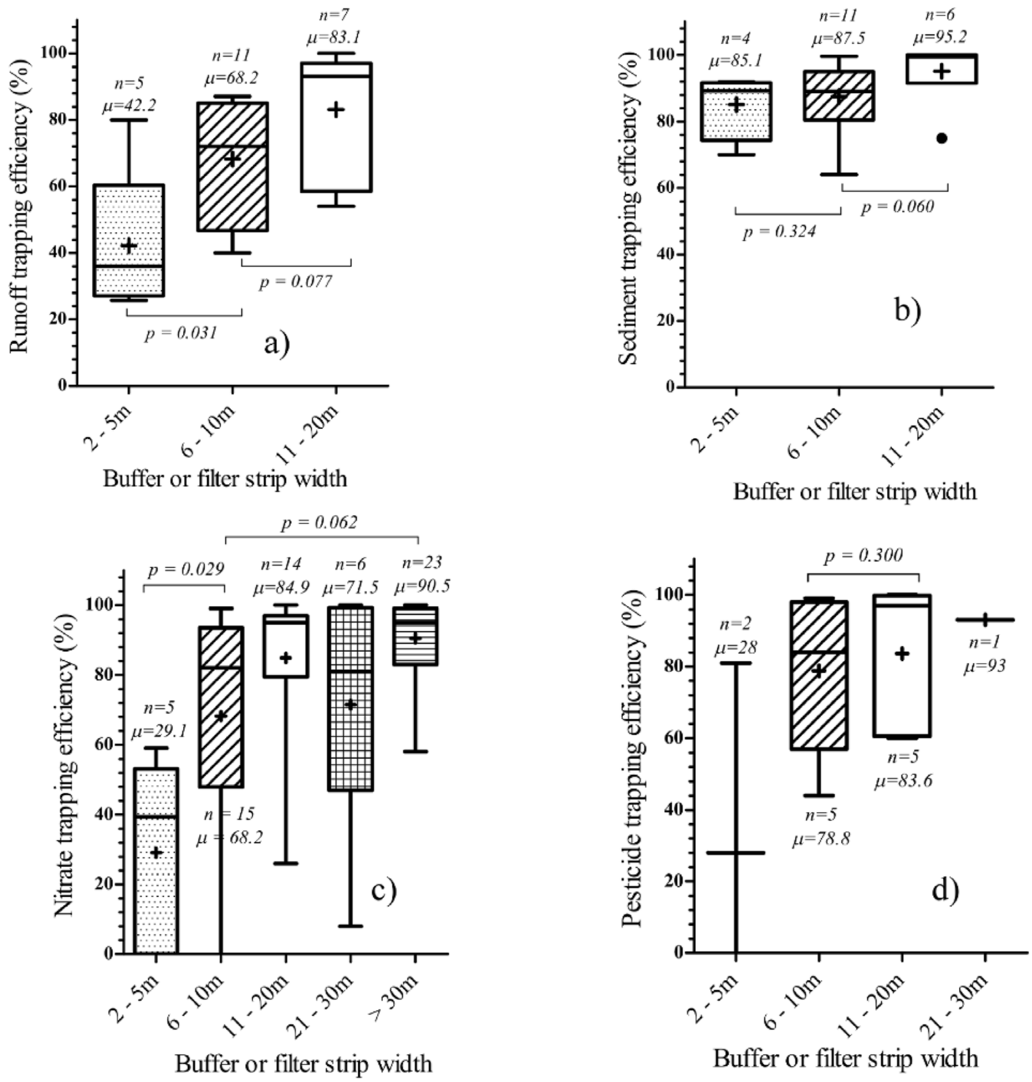


Fig. 4. Trapping efficiencies of vegetated filter strips in an Illinois watershed for a) runoff, b) sediment, c) nitrate, and d) pesticides [3].

valid if these areas are more profitable when growing row crops than biomass crops. However, if biomass in filter strips can be harvested and sold with lower management costs, then planting them would provide an economic benefit. The Billion Ton 2016 vol 1 report [43] analyzed future scenarios that incorporated transportation and logistic costs to the final feedstock prices. In this analysis, feedstocks were assumed to be transformed into intermediate products at a regional processing facility before being converted to biofuel, biopower, or bioproducts at a biorefinery [43]. They assume in the future a feedstock supply system that includes a preprocessing depot to aggregate multiple feedstock sources and multiple end uses that could transform raw biomass into a stable commodity suitable for long-distance transport and handling in existing infrastructure [43]. Economic scenario projections were run using the POLYSYS model and, given the previous assumptions, an estimated 217 million tons of biomass could be available at a delivered cost less than $84 per ton (assuming a roadside/farm gate price of $60/ton) [43]. Farmers may decide not to plant riparian buffers if they interfere with field operations or occupy land that can be more profitably planted in food crops. Given sufficient distance from water bodies, riparian buffers can be harvested for biofuels without negatively impacting water quality. One study in Illinois estimated that planting a 0.32 ha saturated buffer of willow on a 2.63 ha corn field would reduce nitrate loss by up to 61% annually and have a similar cost to other best management practices [44], while providing another income stream to the farmer.

On average, income increased by 43% when the filter strip of switchgrass, little bluestem, big bluestem, prairie cordgrass, and cup plant (*Silphium perfoliatum*) was harvested as an energy feedstock compared with a situation in which farmers only collected income from water quality incentives under the US Department of Agriculture (USDA) Farm Services Agency Conservation Reserve Program (CRP) [45]. Tyndall [29] estimated annual costs of contour prairie strips at $590 to $865 per hectare based on a 15-year average. These costs included site preparation, establishment, maintenance, and lost revenue due to land-use change. Costs were reduced by 85% if the strips were enrolled in the CRP. Such government programs that pay farmers to maintain buffers as part of a wildlife-friendly landscape help to improve the 'bottom line' for farmers. In SWAT simulations of the Iowa river, buffers had significant water quality benefits, but only minimal impacts on corn and soy residue harvest [22], which is important to note in an economic context. In a similar study using the SWAT model, 30-m buffers were simulated around rivers of the South Fork watershed, Iowa. The buffers covered 1,508 ha and would, if harvested, produce around 12,442 dry metric tons of feedstock [46]. A conversion of 80 gallons per dry ton can be used to estimate potential fuel production of 2.7 ML [46].

Climate extremes have recently caused delays in planting of corn and soy in the central US, and may influence economic trade-offs

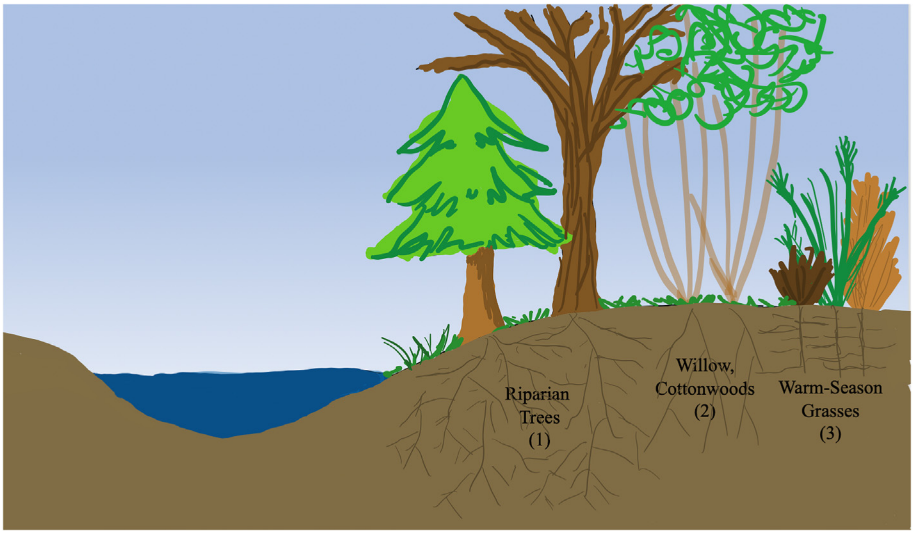


Fig. 5. An example of a multi-tiered riparian buffer. The first tier (1) is composed of riparian trees and understory shrubs with fast-growing roots that tolerate wet conditions. The second tier (2) is composed of trees that tolerate occasional flooding and in this diagram are represented by willow and cottonwoods which can be harvested for biomass. The third and final tier (3) is comprised of warm-season grasses.

associated with filter-strip placement. The risk of flooding is lower when a filter strip is positioned to intercept moderate runoff volumes than when it is positioned to intercept high-storm flows. On fields with low slopes, it is more important to locate buffer strips at the edges of fields because they do not interfere with field drainage, which can impair agricultural productivity [36]. Biomass feedstocks include perennial species that are better adapted than annual crops to wet soils (see Section 3.1). For example, biomass production by a mixture of native species was compared in the Prairie Pothole region of the US [45]. Crop mixtures including several forbs, C3 grasses, and three species of C4 grass [prairie cordgrass (*Spartina pectinate*), big bluestem (*Andropogon gerardii*), and little bluestem (*Schizachyriuym scoparium*)]. Along a gradient from xeric to mesic conditions, C4 species performed better at the extremes and, therefore, produced harvestable biomass more-consistently than switchgrass grown in a monoculture [45], especially at the extremes of switchgrass topographic range.

2.2.3. Decisions that meet multiple objectives

Multi-tiered (e.g., three-zone) riparian buffers (Fig. 5) can be designed to serve multiple purposes [47]. Closest to rivers, streambanks are stabilized by trees and understory shrubs with fast-growing roots that tolerate wet conditions (Fig. 2). To provide shading, buffers should be at least 10-m wide and tree height is at least half the width of the stream. Trees on the streambank, when they fall, also supply large woody debris, which plays a role in forming pool-riffle sequences, providing allochthonous energy sources, and aquatic habitat. The second tier is wider and comprised of trees that tolerate occasional flooding. This tier serves the functions of lowering floodwater, removing nutrients, and degrading pesticides. Additionally, this tier can be managed for biomass by growing such species as willow, cottonwoods (*Populus L.*), or oil-producing trees (e.g., hazelnut). A third tier, furthest from the stream and along the edge of the field, should be at least 7-m wide and planted in switchgrass or other warm-season grasses [47]. This tier can also be planted with forbs, providing additional benefits to pollinators [48,49]. Sullivan [50] surveyed residents, farmers, and academics from central Illinois (USA) on preference for 2- versus 3-tier buffers. Surveyed individuals showed that local non-farmer residents and researchers preferred 3-tier buffers more than farmers did. Farmer concerns included economic losses, increased maintenance, and potential for problematic wildlife. Research to understand and address these concerns is needed.

3. Spatial decisions about artificial drainage systems

Artificial drainage systems, or tile drains, are a series of underground pipes that remove excess water from soil subsurface and lower soil moisture levels. Tile drainage is widespread in the central US. In landscapes drained by artificial drainage systems, water from networks of drains are collected into larger agricultural drainage districts. Artificial drainage makes it possible to cultivate depressions with saturated soils for traditional row crops that are not tolerant of wet conditions. In addition, draining wet fields enables farmers to use equipment earlier in spring. Prior to placing farmland on the market, new drains are installed to guarantee that the land can be insured.

Tile drains have adverse effects on water quality of runoff because they short-circuit infiltration of water into soil. Tile flow concentrates soluble nitrogen, phosphorus, and to a lesser-extent sediment and sediment-bound phosphorus in runoff [51,52]. However, targeted placement of conservation features on tile-drained lands can ameliorate water quality impacts of tiles [53,54].

Below, we discuss two types of spatial decisions related to tile-drain management at both the watershed- and field-scale: 1) where tile-drains are needed for agricultural and bioenergy crops, and 2) where mitigation options (e.g., increased tile spacing or depth, plugging drains, nitrogen abatement, controlled drainage, wetlands) can be deployed to minimize adverse effects of drainage on water quality (Fig. 6).

3.1. Watershed-scale decisions

At the watershed scale, the main considerations are slope and drainage district boundaries. Tile drains are not needed on lands with higher slopes. Evans [55] found that artificial drainage was not needed to improve production of annual crops in lands with slopes greater than 0.5%. A study of the Arkansas-White-Red river basin found little benefit for residue yield or water quality associated with simulating tiling on slopes exceeding 1% [22]. Similarly, substantial reductions in nitrate (> 28%) from corn-soy dominated landscapes in the Iowa River basin when tiles were assumed to be plugged on lands with slopes less than 2%, and a similar reduction was simulated when tiles were plugged on lands with less than 1% slope. In these simulations using the SWAT model, tiling had minimal effects on crop residue yield, sediment loadings, or phosphorus loadings [22].

Where tile drainage is common, drainage districts are a relevant intermediate scale when planning mitigation [56]. Placing constructed wetlands or catch basins at the collection point where tile drain systems empty can help to remove nitrate, phosphorus, and sediment (Fig. 2, see Section 4). Nitrate concentrations of tile outflow typically decrease with increasing drainage area because of flow dilution.

Perennial biomass feedstocks, like willow, poplar, miscanthus, and switchgrass, grow better than row crops, such as corn, in saturated soils

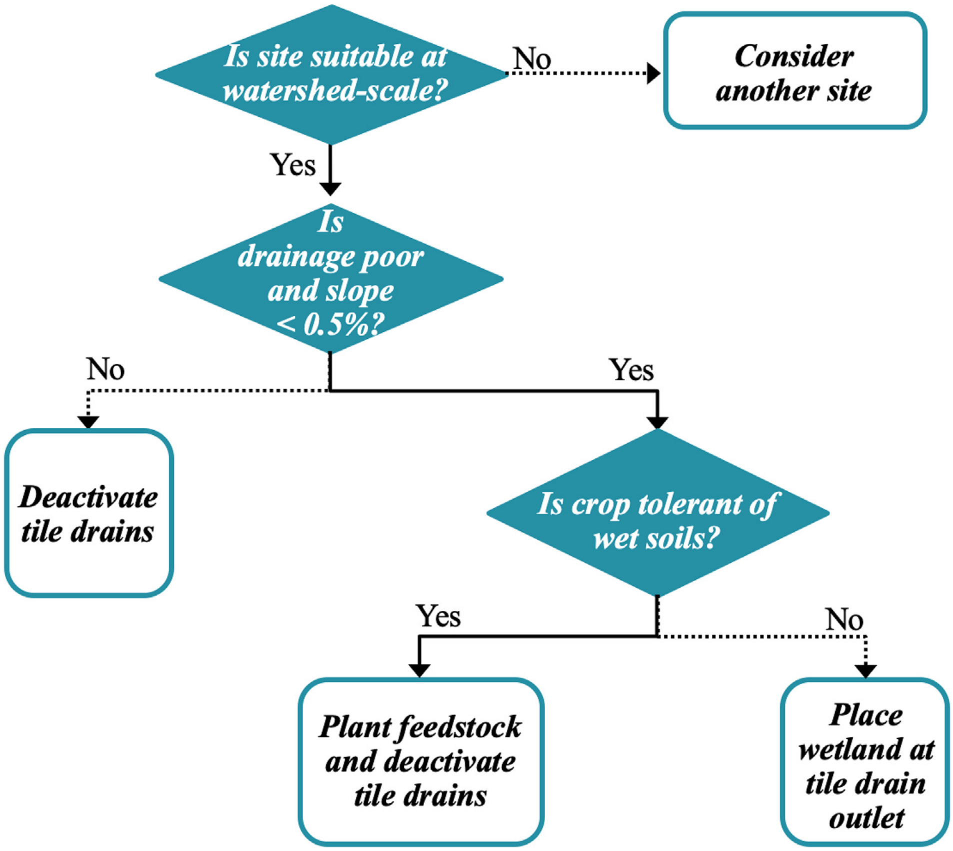


Fig. 6. Decision tree for siting wetlands and tile drains given field attributes.

[57]. Whereas row crops experience lower yields in lowland versus upland positions, prairie grasses, which can be used as biomass feedstocks, perform equally well in saturated lowlands and uplands [57]. The lowland ecotype of switchgrass is tolerant of poorly-drained soils, and is found in streambanks and wetlands [57]. For switchgrass, establishment rates were higher under wet-soil conditions than on well-drained sites; in post-establishment years, yields did not increase as they did on well-drained sites [58]. Likewise, short-rotation woody crops (e.g., poplar and willow) are early-successional species that are adapted to riparian floodplains and seasonal flooding to germinate. Willow and cottonwood out-performed row crops and grasses in lowland position [59]. These biomass feedstocks have less need for tile drainage than annual crops. Thus, the potential for deactivating tile drains without a yield penalty is higher for lands growing perennial bioenergy crops than for traditional commodity crops. Therefore, we predict that spatial designs that place perennial crops in lowland position are likely to produce better outcomes and decrease the need for artificial drainage.

3.2. Field-scale decisions

Field-scale knowledge is essential for introducing effective management practices to improve water quality. Although watershed-scale decisions are important to consider, in-field techniques, such as fertilizer timing, application method, application rate, and cover crops, may have the best chance for broad implementation and may reduce nitrate losses before they reach the tile line [60]. In determining the location of tile drains, the primary concern is where to mitigate for the use of tiles to benefit agricultural yields and improve water quality (Fig. 6). Besides location of tile drains in a field, there are other mitigation options associated with tile drains at the field scale. These include 1) plugging tile drains or otherwise reducing drainage intensity and 2) increasing the precision of fertilizer application to reduce fertilizer losses near tile drains.

Tile removal is an important mitigation option that can be achieved by crushing sections of tile (tile breaks) and plugging the remaining pipe using concrete [51]. A policy analysis found that the high cost of plugging tile drains precludes either continued production or land retirement from being feasible for nitrogen-abatement [51]. Reduced yields on non-drained land added to the potential cost to farmers of plugging tile drains [51]. However, reducing the simulated cost of plugging from $200 per acre to $20 per acre made plugging economically feasible [51]. For perennial biomass crops, plugging tiles is even more feasible because plants are more tolerant of wet soils.

Drainage intensity (i.e., spacing and depth of tile drains) can be reduced to diminish the downstream impact of nutrient-laden tile flow. Soil properties and local weather conditions should be taken into account when setting the depth and spacing of drains to ensure that the system provides only the minimum drainage intensity needed for the crop [61]. Plugging some drain inlets can effectively reduce drain spacing. Kladivko [62] emphasized that drain flow and nitrate-nitrogen losses per unit area were greater for narrower spacing. However, other studies have found phosphorus concentration was influenced by depth, but not by spacing [52]. Burying tile drain lines at greater depths prevents sediment- and nutrient-laden runoff entering drains after precipitation events [52].

Adjusting the amount and timing of fertilizer applied to increase plant uptake efficiency is more effective when implemented on tile-drained lands than when implemented on non-tile-drained lands. One study evaluated the cost-effectiveness of nitrogen-abatement policies on two Minnesota watersheds that grew corn, soybean, and switchgrass as feedstocks for bioenergy [51]. The average cost of nitrogen abatement was $0.91 kg^{-1} on tile-drained land and $40.8 kg^{-1} on non-drained land. Thus, reducing fertilizer through precision agriculture near tile drains offers a promising strategy for lowering nitrate leaching through tile drains. Similarly, using formulations without nitrate and timing application to avoid loss of nitrate in runoff is likely to be more important on tile-drained fields.

The timing of fertilizer applied strongly influences leaching of nitrate into tile drains. For example, Nangia [63] investigated the effect of fertilizer timing and amount on commercial row crop fields in south-central Minnesota. They found a 13% reduction in nitrate losses when the rate of fall fertilizer application was reduced from 180 to 123 kg ha^{-1}. A further 9% reduction in nitrate losses was achieved by simply switching from applying fertilizer in fall to applying fertilizer in spring. Another study monitored tile drain water from Iowa fields in corn-soy rotation [64]. Side-dressing early (the V12 corn growth stage) was found maximize yield and minimize loss of nitrate [64].

4. Spatial decisions about wetlands

Mitsch [65] estimated that nitrate loading to the Gulf of Mexico could be reduced by 300,000 to 800,000 dry metric tons per year (1.4%–3.8%) by creating or restoring wetlands and riparian buffers of perennial grasses on 0.7–1.8% of the land in the Mississippi River Basin. Wetlands (e.g., swamps, marshes, floodplains, and bogs) improve water quality of through-flowing water by retaining and transforming nutrients, sediment, metals, and some contaminants [66]. In addition, they sequester carbon and support the most biologically diverse of ecosystems [67,68]. Wetlands are responsible for significant reductions in nitrate through denitrification.

Like natural wetlands, constructed wetlands reduce the nutrient load of through-flowing water by removing nitrate and phosphorus from surface and subsurface runoff [69]. In tile-drained landscapes, restoration involves disabling drainage systems and installing ditch plugs (an earthen wall used to impound water) to prevent drainage through tiles. Previously-drained wetlands can be restored by planting vegetation tolerant of mesic conditions on lands surrounded by berms.

4.1. Watershed-scale decisions

Large wetlands retain water longer and therefore produce higher-quality effluent. Water parcels are exposed to biogeochemical processes (such as denitrification) longer and soil particles have more opportunity to settle [70]. One guideline is that wetlands contribute significantly to water-quality improvement if they account for at least 2–7% of watershed area [69]. However, another study demonstrated that the spatial distribution of required wetland area is a function of peak-flow capture [71]. More, small wetlands located in tributary basins can achieve the same water quality improvements as a few, large wetlands downstream [71]. The design principle is to protect downstream water quality by ensuring that peak flows can be stored in all nested basins (Fig. 6).

In addition to water retention, slope, soil type, pH, and other attributes have significant influence on a wetlands ability to improve water quality [30] (Fig. 6). Wetland functions are often supported by groundwater connection, and they are best situated where the water table is shallow and/or in areas with poorly-drained hydric soils. However, soils with a pH lower than five should be avoided as these are more likely to emit nitrous oxide via incomplete denitrification [67]. To improve water degraded by agriculture, constructed wetlands should be placed where the watershed tends to collect water so that runoff will be intercepted by the wetland before entering waterways. Incorporating wetlands into tile-drained watersheds will have a positive synergistic effect on water quality in agricultural watersheds if they are placed at the outlets of drains [72].

4.2. Field-scale decisions

In watersheds dominated by agriculture, wetlands covered by a well-structured plant community are most successful in retaining nutrients. Wetlands with perennials such as cattails (*Typha*), reeds (*Phragmites*), and rushes (*Juncus*) improve water quality by increasing water residence time [73]. Biomass crops, such as cattails, reeds, sugarcane, switchgrass, and prairie cordgrass, provide the type of vegetation structure that is needed to increase wetlands retention abilities [30,71,74]. Wetlands with dense vegetation and flat bottoms support higher rates of denitrification. From a biofuel perspective, mixtures of switchgrass and prairie cordgrass are high-yielding wetland species in the tallgrass prairie region, with the latter producing a yield of 13.2 Mg ha^{-1} when grown as monocultures [75]. In a Portuguese study, sugarcane (*Saccharum officinarum*) was successfully grown in a constructed wetland subject to seasonal flooding [76]. Average nutrient removal efficiencies of 77% ± 4% for TP and 60% for ± 12% for TN were recorded [76]. Cultivating row crops in river floodplains can result in lower crop yields and lost top soil. Harvest of perennials adapted to wet-soil conditions for biofuels may be a more-sustainable alternative [75,77]. Furthermore, careful management of constructed wetlands may help to promote conservation of these important ecosystems and the ecosystem services that they provide [78].

5. Spatial decisions about residue removal

Decisions about residue removal at the field scale consider both water quality and farmer profit. Agricultural residues can help the U.S. meet bioenergy targets [79]. However, removing too much crop residue can degrade the long-term productive capacity of soil resources by depleting soil organic matter [80,81]. Spatially variable guidelines for residue removal can be formalized through spatial optimization [79]. For each spatial unit, crop residue removal rates that minimize environmental impacts while maximizing biomass feedstock are estimated [82]. Factors affecting the optimal harvest rate include biophysical characteristics of the field, management practices, and environmental goals [83].

5.1. Watershed-scale decisions

Cibin [84] conducted a field-scale optimization and determined that optimal removal rates were evenly distributed among various fields within a watershed containing corn, miscanthus, and switchgrass. However, the watershed-scale optimization seeking to maximize water quality recommended a decrease in harvested corn stover from fields closer to the watershed outlet and increased quantities 40–50 km away from the outlet [84]. It should be noted that meeting other objectives, such as maintaining soil fertility, will likely depend on local soil conditions, and not proximity to the outlet [82,85].

5.2. Field-scale decisions

At the field-scale, modern hand-held and tractor-based precision agriculture applications make it possible to map a field and optimize residue removal. The precision agriculture tool, AgSolver, was designed to help farmers tailor removal rates to local conditions and to identify unprofitable areas of a field [82]. Such tools make it possible to remove more corn stover from areas with relatively low risk of soil erosion and nutrient losses, such as flatter areas and areas where conservation tillage or no-till practices are used [86]. Studies indicate that higher crop-residue removal is possible at locations with lower risk of erosion (e.g. fields with no-till or conservation tillage; field with slope < 2%) [87–90]. Guidelines suggest between 30 and 50% removal rate of residue [89]. Subfield-scale variability in soil properties (e.g., slope, texture, and other organic matter content), tillage, and crop rotation that affect grain yield significantly affect the amount of residue that can be sustainably removed from different areas within a single field [90]. Determining sustainable removal rates for agricultural residues requires assessing multiple agronomic and environmental factors

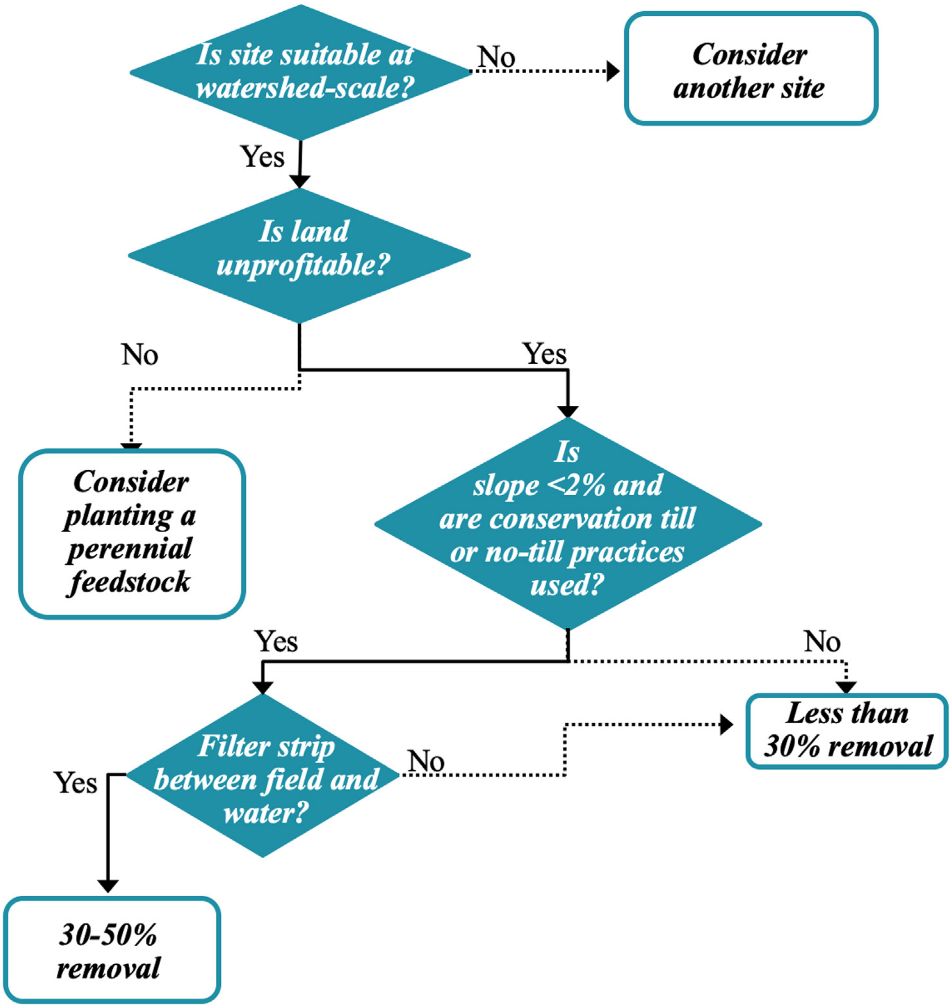


Fig. 7. Decision tree for placing residue removal rates based on field attributes. Removal rates from Ref. [89].

simultaneously, including soil and land management practices [91] (Fig. 7).

In summary, optimization has been used to determine context-specific optimal crop residue removal rates. Recommendations of higher levels of residue removal without adverse water quality impacts include removing more from areas away from outlets, with low slope. In addition, higher removal rates are possible when using conservation tillage and where a filter strip has been planted between field and water.

6. Interactions among spatial decisions

Spatial decisions on land management are not made independently, and joint consideration of spatial decisions can lead to significantly improved water-quality outcomes. Interactions between tile drains, wetlands, tile drains, filter strips, and residue removal rates are described below. These are summarized in Fig. 8.

Use of one mitigation practice can reduce the need for others, thereby increasing farmer profit without a water quality cost. For example, residue removal rates can be higher where filter strips and grassed waterways reduce pollutant load from agricultural fields to water bodies [36]. The presence of tile drains can also influence where filter strips are located to be effective. Potholes are appropriate targets for filter strips because subsurface leaching occurs at these sites, especially when tiles are installed [92]. On relatively flat fields, it is more important to locate filter strips at the edges of fields because they do not interfere with field drainage, which can impair agricultural productivity [93]. Additionally, filter strips can be used in parallel with wetlands to reduce pesticide loads [94]. This is a particularly useful strategy if there is a space limitation on constructed mitigation practices.

Artificial drainage from poorly drained agricultural soils is known to increase nitrate and phosphorus losses [52]. These can be reduced by strategic placement of wetlands in the landscape [95]. Outflow points represent the ideal spatial opportunity to treat nutrient-rich discharge. For example, wetlands can intercept and treat drainage from tile-drains before being released to downstream water bodies or subsurface aquifers [71]. One optimization study found that it was not possible to eliminate phosphorus loadings by using only riparian buffers, but that it was also necessary to divert drainflow, which can be rich in phosphorus [96].

Residue removal increases soil erosion because less residue is available to protect the ground from rainfall impact, runoff, and wind forces. Using filter strips on fields where crop residues are removed can decrease the amount of soil erosion that occurs [97]. Similarly, constructing wetlands for fields where residue removal occurs can benefit water quality and increase harvest of residues and minimize impact on water quality.

7. Conclusion

This review summarized context-specific placement decisions based on rules of thumb derived from spatial optimization models to address four types of decisions: spatial decisions about vegetated filter strips, artificial drainage, wetlands, and residue removal. Results addressed

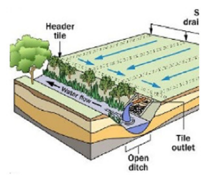
	Filter strips	Tile drains	Wetland	Residue removal
Filter strips		Both nitrate and sediment-bound phosphate are remediated.	Filter strips favor denitrification and benefits can be amplified when combined with wetlands.	Higher residue harvest can occur without an increase in soil erosion.
Tile drains	Locate strips at the edges of fields or at outlets where they do not interfere with field drainage.		Crop productivity can be increased without a nitrate penalty when wetlands are located at tile drain outlets.	Residue harvest can occur on a tile-drained field without a significant change in nutrient concentration of discharge.
Wetland	Can be used in parallel to reduce pesticide loads.	Placing tile drains upstream of wetlands can reduce the nutrient loading in discharges downstream.		Harvest for biofuels can be increased without adverse impacts on water quality.
Residue removal	Harvest for biofuels can be increased without adverse impacts on water quality.	Surface runoff is concentrated in tile drainage which without treatment will negatively affect water quality.	Harvest for biofuels can be increased without adverse impacts on water quality.	

Fig. 8. Potential interactions among conservation decisions can lead to increased ecosystem services.

placement covariates at different scales. This is important because biophysical processes are dominated by different spatially variable attributes (covariates) at different spatial scales [98]. At local scales, processes influenced by spatial proximity dominate. However, as we scale up, spatial location, by itself, and proximity, become meaningless. At these scales, spatial variation in covariates inform spatial decisions, not location *per se*. Models built on non-spatial covariates can transcend the sites at which they are developed (i.e., they are transferable). Therefore, our review identified spatial covariates relevant at watershed and field scales. We highlighted important interactions, offered conditional rules for co-location, and suggested ways to formalize future optimization studies to refine these rules. In section 6, we described synergistic effects that can occur when implementing multiple conservation practices. We found that combining conservation practices in a way that accounts for their interactive effects can significantly improve water quality outcomes. Based on our review, we determine that by making spatial decisions on conservation practices, bioenergy landscapes can be designed to improve water quality and enhance other ecosystem services.

Funding

Funding for this research was provided by the Bioenergy Technologies Office of the US Department of Energy. We appreciate support from Kristen Johnston, manager for BETO's Sustainability program. This manuscript has been co-authored by UT-Battelle, LLC under Contract No. DE-AC05-00OR22725 with the U.S. Department of Energy. The US government retains and the publisher, by accepting the article for publication, acknowledges that the US government retains a non-exclusive, paid-up, irrevocable, world-wide license to publish or reproduce the published form of this manuscript, or allow others to do so, for US government purposes. DOE will provide public access to these results of federally sponsored research in accordance with the DOE Public Access Plan (http://energy.gov/downloads/doe-public-access-plan).

Acknowledgements

The authors would like to thank the anonymous reviewers for helpful comments.

References

[1] Millennium Ecosystem Assessment, Ecosystems and Human Well-Being: Biodiversity Synthesis, World Resources Institute, Washington, DC, 2005, p. 86.
[2] J.G. Hof, M. Bevers, Spatial Optimization for Managed Ecosystems, Columbia University Press, New York, 1998.
[3] H. Ssegane, M.C. Negri, J. Quinn, M. Urgun-Demirtas, Multifunctional landscapes: site characterization and field-scale design to incorporate biomass production into an agricultural system, Biomass Bioenergy 80 (2015) 179–190.
[4] H. Blanco-Canqui, Growing dedicated energy crops on marginal lands and ecosystem services, Soil Sci. Soc. Am. J. 80 (2016) 845–858.
[5] D. Allan, D. Erickson, J. Fay, The influence of watershed land use on stream integrity across multiple spatial scales, Freshw. Biol. 37 (1) (1997) 149–161.
[6] K. Jensen, C.D. Clark, P. Ellis, B. English, J. Menard, M. Walsh, D. Torre Ugarte, Farmer willingness to grow switchgrass for energy production, Biomass Bioenergy 31 (2007) 773–781.
[7] W. Jiang, K.Y. Zipp, M. Jacobson, Economic assessment of landowners' willingness to supply energy crops on marginal lands in the northeastern of the United States, Biomass Bioenergy 113 (2018) 22–30.
[8] O. Joshi, S.R. Mehmood, Factors affecting nonindustrial private forest landowners' willingness to supply woody biomass for bioenergy, Biomass Bioenergy 35 (2011) 186–192.
[9] E. Brandes, G. McNunn, I. Schulte, I. Bonner, D. Muth, B. Babcock, b. Sharma, E. Heaton, Subfield profitability analysis reveals an economic case for cropland diversification, Environ. Res. Lett. 11 (2016) 1–13.
[10] R.A. Efroymson, M.H. Langholtz, K. Johnson, B. Stokes, C.C. Brandt, M.R. Davis, ...

J. Dunn, 2016 billion-ton report: advancing domestic resources for a thriving bioeconomy, Environmental Sustainability Effects of Select Scenarios from Volume 1, vol. 2, Oak Ridge National Laboratory (ORNL), Oak Ridge, TN (United States), 2017.
[11] M.J. Helmers, X. Zhou, H. Asbjornsen, R. Kolka, M.D. Tomer, R.M. Cruse, Sediment removal by prairie filter strips in row-cropped ephemeral watersheds, J. Environ. Qual. 41 (2012) 1531–1539.
[12] C. Pan, L. Ma, Z. Shangguan, Effectiveness of grass strips in trapping suspended sediments from runoff, Earth Surf. Process. Landforms 35 (9) (2010) 1006–1013.
[13] M.A. Nearing, L.D. Norton, X. Zhang, Soil erosion and sedimentation, in: W.F. Ritter, A. Shirmohammadi (Eds.), Agricultural Nonpoint Source Pollution-Watershed Management and Hydrology, Lewis, Washington D.C., 2001, pp. 29–58.
[14] M.D. Tomer, M.G. Dosskey, M.R. Burkart, D.E. James, M.J. Helmers, D.E. Eisenhauer, Methods to prioritize placement of riparian buffers for improved water quality, Agrofor. Syst. 75 (2009) 17–25.
[15] H.G. Momm, R.L. Bingner, Y. Yuan, M.A. Locke, R.R. Wells, Spatial characterization of riparian buffer effects on sediment loads from watershed systems, J. Environ. Qual. 45 (5) (2014) 1736–1753.
[16] J.M. Sheridan, R. Lowrance, D.D. Bosch, Management effects on runoff and sediment transport in riparian forest buffers, Am. Soc. Agric. Eng. 42 (1) (1999) 55–64.
[17] R. Lowrance, L.S. Altier, J.D. Newbold, R.R. Schnabel, P.M. Groffman, J.M. Denver, D.L. Correll, J.W. Gilliam, J.L. Robinson, R.B. Brinsfield, K.W. Staver, W. Lucas, A.H. Todd, Water quality functions of riparian forest buffers in chesapeake bay watersheds, Environ. Manag. 21 (5) (1997) 68–712.
[18] W.K. Dodds, R.M. Oakes, Headwater influences on downstream water quality, Environ. Manag. 41 (2008) 367–377.
[19] J. Wu, K. Skelton-Groth, Targeting conservation efforts in the presence of threshold effects and ecosystem linkages, Ecol. Econ. 42 (2002) 313–331.
[20] M. Jha, K. Schilling, P. Gassman, C. Wolter, Targeting land-use change for nitrate-nitrogen load reductions in an agricultural watershed, J. Soil Water Conserv. 65 (2010) 342–352.
[21] M. Ha, M. Wu, Simulating and evaluating best management practices for integrated landscape management scenarios in biofuel feedstock production, Biofuels Bioprod. Biorefining 9 (2015) 709–721.
[22] H.I. Jager, M. Wu, M. Ha, L. Baskaran, J. Kreig, Water quality responses to simulated management practices on agricultural lands producing biomass feedstocks in two tributary basins of the Mississippi river. Pg 139-178, in: R. Efroymson, K. Johnson, M. Langholtz (Eds.), 2010 Billion-Ton Report (BT16) Volume 2: Environmental Sustainability Effects of Select Scenarios from Volume 1, Oak Ridge National Laboratory and the Department of Energy Bioenergy Technologies Office, Oak Ridge, TN, 2017.
[23] Y. Li, K. Guan, G.D. Schnitkey, E. DeLucia, B. Peng, Excessive rainfall leads to maize yield loss of a comparable magnitude to extreme drought in the United States, Glob. Chang. Biol. 25 (2019) 2325–2337.
[24] G. Gopalakrishnan, M.C. Negri, S.W. Snyder, A novel framework to classify marginal land for sustainable biomass feedstock production, J. Environ. Qual. 40 (2011) 1593–1600.
[25] O. Evrard, K. Vandaele, B. Van Wesemael, C.L. Bielders, A grassed waterway and earthen dams to control muddy floods from a cultivated catchment of the Belgian loess belt, Geomorphology 100 (2008) 419–428.
[26] K. Vandaele, J. Lammens, P. Priemen, E. Evrard, How to control muddy floods from cultivated catchments, Lessons from the Melsterbeek catchment in Flanders (Belgium), Samenkering Land en water, St-Truiden, Belgium, 2013.
[27] E.S. Parish, M.R. Hilliard, L.M. Baskaran, V.H. Dale, N.A. Griffiths, P.J. Mulholland, A. Sorokine, N.A. Thomas, M.E. Downing, R.S. Middleton, Multimetric spatial optimization of switchgrass plantings across a watershed, Biofuels Bioprod. Biorefining 6 (2012) 5872.
[28] M. Geza, B. Barfield, R. Huhnke, A. Stoecker, D. Storm, E. Stevens, Comparison of targeted replacement and vegetative filter strips for sediment control and cost effectiveness, J. Water Resour. Plan. Manag. 135 (2009) 406–409.
[29] J.C. Tyndall, L.A. Schulte, M. Liebman, M. Helmers, Field-level financial assessment of contour prairie strips for enhancement of environmental quality, Environ. Manag. 52 (2013) 736–747.
[30] D. Moreno-Mateos, U. Mander, F.A. Comin, C. Pedrocchi, E. Uuemaa, Relationships between landscape pattern, wetland characteristics, and water quality in agricultural catchments, J. Environ. Qual. 37 (2008) 2170–2180.
[31] M.G. Dosskey, D.E. Eisenhauer, M.J. Helmers, Establishing conservation buffers using precision information, J. Soil Water Conserv. 60 (2005) 349–354.
[32] X. Liu, X. Zhang, M. Zhang, Major factors influencing the efficacy of vegetated buffers on sediment trapping: a review and analysis, J. Environ. Qual. 37 (2008) 1667–1674.
[33] R.A. Fischer, J.C. Fischenich, Design Recommendations for Riparian Corridors and Vegetated Buffer Strips, DTIC Document, 2000.
[34] M.G. Dosskey, M. Helmers, D.E. Eisenhauer, A design aid for determining width of filter strips, J. Soil Water Conserv. 63 (2008) 232–241.
[35] M.A. Harris, G. Iyer, Small changes, big impacts: prairie conservation strips, Natural Resource Ecology and Management Publications and Papers, 2014, pp. 1–4.
[36] S. Dabney, M. Moore, M. Locke, Integrated management of in-field, edge-of-field, and after-field buffers, J. Am. Water Resour. Assoc. (2006) Paper No. 04201.
[37] M. Sahu, R.R. Gu, Modeling the effects of riparian buffer zone and contour strips on stream water quality, Ecol. Eng. 35 (2009) 1167–1177.
[38] G. Bentrup, Conservation Buffers: Design Guidelines for Buffers, Corridors, and Greenways, Department of Agriculture, Forest Service, Southern Research Station, Asheville, NC, 2008, p. 110.
[39] R. Muoz-Carpena, J.E. Parsons, A design procedure for vegetative filter strips using VFSMOD-W, Trans. ASAE 47 (2004) 1933–1941.
[40] R. Lowrance, L. Altier, R. Williams, S. Inamdar, J.M. Sheridan, D.,. Bosch, D. Thomas, REMM: the riparian ecosystem management model, J. Soil Water Conserv. 55 (2000) 27–34.
[41] M.T. Brown, Buffer Zones for Water, Wetlands, and Wildlife: A Final Report on the Evaluation of the Applicability of Upland Buffers for the Wetlands of the Wekiva Basin, Center for Wetlands, University of Florida, 1987.
[42] R.D. Barling, I.D. Moore, Role of buffer strips in management of waterway pollution: a review, Environ. Manag. 18 (1994) 543–558.
[43] U.S. Department of Energy, 2016 Billion-Ton Report: advancing domestic resources for a thriving bioeconomy, in: M.H. Langholtz, B.J. Stokes, L.M. Eaton (Eds.), Economic Availability of Feedstocks, Vol 1 Oak Ridge National Laboratory, Oak Ridge, TN, 2016, , https://doi.org/10.2172/1271651.
[44] J.F. Cacho, M.C. Negri, C.R. Zumpf, P. Campbell, Introducing Perennial Biomass Crops into Agricultural Landscapes to Address Water Quality Challenges and Provide Other Environmental Services vol. 7, Wiley Interdisciplinary Reviews-Energy and Environment, 2018.
[45] C.J. Zilverberg, K. Teoh, A. Boe, W.C. Johnson, V. Owens, Strategic use of native species on environmental gradients increases diversity and biomass relative to switchgrass monocultures, Agric. Ecosyst. Environ. 215 (2016) 110–121.
[46] M. Ha, M. Wu, Land management strategies for improving water quality in biomass production under changing climate, Environ. Res. Lett. 12 (2017) 034015.
[47] P. Bongard, G. Wyatt, U.O.M. Extension (Ed.), Benefits of Riparian Forest Buffers, 2010, p. 2.
[48] F. Kohler, J. Verhulst, R. Van Klink, D. Kleijn, At what spatial scale do high-quality habitats enhance the diversity of forbs and pollinators in intensively farmed landscapes? J. Appl. Ecol. 45 (2007) 753–762.
[49] L. Kutt, T. Paal, K. Lohmus, I.J. Rammi, K. Zobel, J. Liira, Multi-user quality of floral services along a gradient of margin habitats between semi-natural grasslands and forests, Appl. Veg. Sci. 21 (2018) 363–372.
[50] W.C. Sullivan, O.M. Anderson, S.T. Lovell, Agricultural buffers at the ruralurban fringe: an examination of approval by farmers, residents, and academics in the Midwestern United States, Landsc. Urban Plan. 69 (2004) 299–313.
[51] D.R. Petrolia, P.H. Gowda, Missing the boat: midwest farm drainage and Gulf of Mexico hypoxia, Rev. Agric. Econ. 28 (2006) 240–253.
[52] K.W. King, M.R. Williams, N.R. Fausey, Contributions of systematic tile drainage to watershed-scale phosphorus transport, J. Environ. Qual. 44 (2015) 486–494.
[53] A.M. Lemke, K.G. Kirkham, T.T. Lindenbaum, M.E. Herbert, T.H. Tear, W.L. Perry, J.R. Herkert, Evaluating agricultural best management practices in tile-drained subwatersheds of the mackinaw river, Illinois, J. Environ. Qual. 40 (2011) 1215–1228.
[54] B.A. Needelman, P.J. Kleinman, J.S. Strock, A.L. Allen, Drainage ditches improved management of agricultural drainage ditches for water quality protection: an overview, J. Soil Water Conserv. 62 (2007) 171–178.
[55] R.O. Evans, R.W. Skaggs, J.W. Gilliam, Controlled versus conventional drainage effects on water quality, J. Irrig. Drain. Eng. 121 (1995).
[56] C.D. Ikenberry, M.L. Soupir, K.E. Schilling, C.S. Jones, A. Seeman, Nitrate-nitrogen export: magnitude and patterns from drainage districts to downstream river basins, J. Environ. Qual. 43 (2014) 2024–2033.
[57] A.C. Von Haden, M.E. Dornbush, Prairies thrive where row crops drown: a comparison of yields in upland and lowland topographies in the Upper Midwest US, Agronomy 6 (2016) 32.
[58] J.H. Fike, J.W. Pease, V.N. Owens, R.L. Farris, J.L. Hansen, E.A. Heaton, D.R. Viands, Switchgrass nitrogen response and estimated production costs on diverse sites, Glob. Chang. Biol.: Bioenergy 9 (10) (2017) 1526–1542.
[59] R. Thelemann, G. Johnson, C. Sheaffer, S. Banerjee, H. Cai, D. Wyse, The effect of landscape position on biomass crop yield, Agron. J. 102 (2010) 513–522.
[60] M.B. David, C.G. Flint, L.E. Gentry, M.K. Dolan, G.F. Czapar, R.A. Cooke, T. Lavaire, Navigating the socio-bio-geo-chemistry and engineering of nitrogen management in two Illinois tile-drained watersheds, J. Environ. Qual. 44 (2015) 368–381.
[61] R.W. Skaggs, M.A. Youssef, Effect of drainage water management on water conservation and nitrogen losses to surface waters, 16th National Nonpoint Source Monitoring Workshop, September 14-18, Columbus, Ohio, 2008.
[62] E.J. Kladivko, J.R. Frankenberger, D.B. Jaynes, D.W. Meek, B.J. Jenkinson, N.R. Fausey, Nitrate leaching to subsurface drains as affected by drain spacing and changes in crop production system, J. Environ. Qual. 33 (2004) 1803–1813.
[63] V. Nangia, R.H. Gowda, D.J. Mulla, G.R. Sands, Water quality modeling of fertilizer management impacts on nitrate losses in tile drains at the field scale, J. Environ. Qual. 37 (2008) 296–307.
[64] D.B. Jaynes, Nitrate loss in subsurface drainage and corn yield as affected by timing of sidedress nitrogen, Agric. Water Manag. 130 (2013) 52–60, https://doi.org/10.1016/j.agwat.2013.08.010.
[65] W.J. Mitsch, J.W. Day, J.W. Gilliam, P.M. Groffman, D.L. Hey, G.W. Randall, N.M. Wang, Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: strategies to counter a persistent ecological problem, Bioscience 51 (2001) 373–388.
[66] R.H. Kadlec, Overview: surface flow constructed wetlands, Water Sci. Technol. 32 (1995) 1–12.
[67] M.M. Hefting, R.N. Van Den Heuvel, J.T.A. Verhoeven, Wetlands in agricultural landscapes for nitrogen attenuation and biodiversity enhancement: opportunities and limitations, Ecol. Eng. 56 (2013) 5–13.
[68] L.M. Cowardin, Wetland classification in United States, J. For. 76 (1978) 666–668.
[69] J.T.A. Verhoeven, B. Arheimer, C.Q. Yin, M.M. Hefting, Regional and global concerns over wetlands and water quality, Trends Ecol. Evol. 21 (2006) 96–103.
[70] G.W. Raisin, D.S. Mitchell, The use of wetlands for the control of non-point source pollution, Water Sci. Technol. 32 (1995) 177–186.
[71] M. Babbar-Sebens, R.C. Barr, L.P. Tedesco, M. Anderson, Spatial identification and

optimization of upland wetlands in agricultural watersheds, Ecol. Eng. 52 (2013) 130–142.
[72] C. Lenhart, B. Gordon, J. Gamble, D. Current, N. Ross, L. Herring, ... H. Peterson, Design and hydrologic performance of a tile drainage treatment wetland in Minnesota, Water 8 (2016) 549 USA.
[73] D.C. Pratt, D.R. Dubbe, E.G. Garver, P.J. Linton, Wetland Biomass Production: Emergent Aquatic Management Options and Evaluation, (1984) Report prepared for U.S. Department of Energy Contract No. DE-AC02-83CH100093.
[74] M. Kalcic, I. Chaubey, J. Frankenberger, E. Kladivko, A Geospatial Approach to targeting constructed wetlands for nitrate removal in agricultural watersheds, Appl. Eng. Agric. 28 (2012) 347–357.
[75] C.J. Zilverberg, W.C. Johnson, V. Owens, A. Boe, T. Schumacher, K.,. Reitsma, B. Werner, Biomass yield from planted mixtures and monocultures of native prairie vegetation across a heterogeneous farm landscape, Agric. Ecosyst. Environ. 186 (2014) 148–159.
[76] D.M. Mateus, M.M.N. Vaz, I. Capela, H.J.O. Pinho, The potential growth of sugarcane in constructed wetlands designed for tertiary treatment of wastewater, Water 8 (93) (2016) 1–14.
[77] J.T.A. Verhoeven, T.L. Setter, Agricultural use of wetlands: opportunities and limitations, Ann. Bot. 105 (2010) 155–163.
[78] I. Melts, M. Ivask, M. Geetha, K. Takeuchi, K. Heinsoo, Combining bioenergy and nature conservation: an example in wetlands, Renew. Sustain. Energy Rev. 111 (2019) 293–302.
[79] D.J. Muth Jr., K.M. Bryden, R.G. Nelson, Sustainable agricultural residue removal for bioenergy: a spatially comprehensive US national assessment, Appl. Energy 102 (2013) 403–417.
[80] J. Sheehan, A. Aden, K. Paustian, K. Killian, J. Brenner, M. Walsh, R. Nelson, Energy and Environmental Aspects of using corn stover for fuel ethanol, J. Ind. Ecol. 7 (3–4) (2008) 117–146.
[81] L. Mann, v. Tolbert, J. Cushman, Potential environmental effects of corn (Zea mays L.) stover removal with emphasis on soil organic matter and erosion, Agric. Ecosyst. Environ. 89 (3) (2002) 149–166.
[82] I. Bonner, K. Cafferty, D. Muth, M. Tomer, D. James, S. Porter, D. Karlen, Opportunities for energy crop production based on subfield-scale distribution of profitability, Energies 7 (2014) 6509–6526.
[83] B.M. Gramig, C.J. Reeling, R. Cibin, I. Chaubey, Environmental and economic trade-offs in a watershed when using corn stover for bioenergy, Environ. Sci. Technol. 47 (2013) 17841791.
[84] R. Cibin, Optimal Land Use Planning on Selection and Placement of Energy Crops for Sustainable Biofuel Production, Ph.D. Purdue University, 2013.
[85] I. Bonner, G. Mcnunn, D. Muth Jr., W. Tyner, J. Leirer, M. Dakins, Development of integrated bioenergy production systems using precision conservation and multi-criteria decision analysis techniques, J. Soil Water Conserv. 71 (2016) 182–193.
[86] R. Cibin, I. Chaubey, B. Engel, Simulated watershed scale impacts of corn stover removal for biofuel on hydrology and water quality, Hydrol. Process. 26 (2012) 16291641.
[87] R.M. Cruse, C.G. Herndl, Balancing corn stover harvest for biofuels with soil and water conservation, J. Soil Water Conserv. 64 (2009) 286291.
[88] K.L. Kadam, J.D. McMillan, Availability of corn stover as a sustainable feedstock for bioethanol production, Bioresour. Technol. 88 (2003) 1725.
[89] H. Blanco-Canqui, R. Lal, W.M. Post, L.B. Owens, Changes in long-term no-till corn growth and yield under different rates of stover mulch, Agron. J. 98 (2006) 1128.
[90] D. Muth Jr., K.M. Bryden, A conceptual evaluation of sustainable variable-rate agricultural residue removal, J. Environ. Qual. 41 (2012) 1796–1805.
[91] D. Muth Jr., K.M. Bryden, An integrated model for assessment of sustainable agricultural residue removal limits for bioenergy systems, Environ. Model. Softw 39 (2013) 50–69.
[92] T.B. Moorman, D.B. Jaynes, C.A. Cambardella, J.L. Hatfield, R.L. Pfeiffer, A.J. Morrow, Water quality in Walnut Creek watershed: herbicides in soils, subsurface drainage, and groundwater, J. Environ. Qual. 28 (1999) 35–45.
[93] B. Gharabaghi, R.P. Rudra, P.K. Goel, Effectiveness of vegetative filter strips in removal of sediments from overland flow, Water Qual. Res. J. Can. 41 (2006) 275–282.
[94] E. Passeport, J. Tournebize, C. Chaumont, A. Guenne, Y. Coquet, Pesticide contamination interception strategy and removal efficiency in forest buffer and artificial wetland in a tile-drained agricultural watershed, Chemosphere 91 (2013) 1289–1296.
[95] M.M. Kalcic, J. Frankenberger, I. Chaubey, Spatial optimization of six conservation practices using SWAT in tiledrained agricultural watersheds, J. Am. Water Resour. Assoc. 51 (2015) 956–972.
[96] X. Zhang, X. Liu, M. Zhang, R.A. Dahlgren, M. Eitzel, A review of vegetated buffers and a meta-analysis of their mitigation efficacy in reducing nonpoint source pollution, J. Environ. Qual. 39 (2010) 76–84.
[97] J.K. Newman, A.L. Kaleita, J.M. Laflen, Soil erosion hazard maps for corn stover management using National Resources Inventory data and the Water Erosion Prediction Project, J. Soil Water Conserv. 65 (2010) 211222.
[98] J. Wu, K.B. Jones, H. Li, O.L. Loucks, Scaling and Uncertainty Analysis in Ecology: Methods and Applications, Springer, New York, 2006.