Mapping Contributing Areas for Stormwater Discharge to Streams Using Terrain Analysis

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Abstract

An approach to delineating riparian contributing source areas of stormwater discharge to perennial streams was evaluated. Using publicly available data and software, four variable width investigative buffer models, were constructed using cumulative cost distance calculated over fuzzy set combinations of watershed-relative wetness and stream power for a 10 km first-order Michigan stream. These models were compared to assess the sensitivity of buffer delineation to choice of dynamic versus static wetness index, variable versus uniform soil properties, and flow routing algorithm. These models are grounded in the assumption that riparian segments receiving the greatest discharge have upslope contributing areas dominated by saturated soils and have sufficient stream power for saturated flow to reach the stream. Model inputs consisted of U.S. Geological Survey 10 foot (3 meter) topographic contours and county soil survey digital data. Validation was conducted using data collected during a post storm-event GPS field survey of ponded storm-flow accumulations and concentrated stormflow discharge sites. The implications of these models for targeting water pollution remediation efforts are discussed.

Introduction

Increasing concern about surface water quality has led to the development of best management practices such as grassed waterways, buffer strips, cover crops, and no-till agriculture for mitigating non-point source pollution from sediment and adsorbed pollutants. Effective implementation of these strategies requires an understanding of the processes by which soils, land use/cover, terrain and land management practices influence the detachment and delivery of sediment to surface waters. Geographic information systems (GIS) provide an environment within which such data can be managed and analyzed, and in which formal models of pollutant delivery processes can be formulated and tested. Distributed watershed models (e.g., ANSWERS, AGNPS) are useful for simulating sediment erosion, deposition and delivery to surface waters (e.g., Beasley and Huggins 1982, Young et al. 1989), but incur substantial costs for collecting and compiling the spatial and attribute data needed as inputs (Grayson et al. 1992, João and

Walsh 1992). Even when pollution contributing areas can be located, additional data collection (e.g., ownership, management practices, zoning).is usually required to identify feasible mitigation solutions.

Whether areas of potentially high erosion (i.e., sediment production) are identified using a distributed erosion model or some simpler method (e.g., the Revised Universal Soil Loss Equation), collection of spatially distributed erosion potential and management practice data in support of the mitigation effort can be costly. This chapter describes a terrain analysis approach grounded in hydrological principles that can limit the geographic scope of the data collection effort. A two stage process is suggested. In Stage 1 (Figure 1a), the focus is on collecting and analyzing a limited set of thematic information about an entire watershed (i.e., terrain, drainage channels, and soils). Terrain analysis is then used to identify potential "hot spots" where sediment from potentially erodible source areas might be delivered to the stream channels. We suggest the term investigative riparian buffer for this zone of potential hot spots. Stage 2 entails a data collection effort spatially focused within this investigative riparian buffer, but more expansive in attributes (e.g., land cover, ownership) to support the ranking of potential remediation sites based on a richer information set (Figure 1b). The terrain analysis models presented here should be useful for making the Stage 1 selection.

While a number of authors have suggested that water quality management activities should focus within a set distance from streams (Duda and Johnson 1985, Maas et al. 1985, Phillips 1989, National Research Council 1993), and that such a distance might well vary within a watershed (Walling 1983), the approach described here extends these ideas to the concept of a variable width investigative buffer around stream channels (Hunsaker and Levine 1995). The methods we propose can be carried out with readily obtainable data using well developed hydrological principles embedded in the TAPES-G software to delineate such investigative buffers.

Implications of Sediment for Water Quality

Sediment is the most ubiquitous pollutant of surface waters by both volume and mass (Chesters & Schierow 1985). Eroded soils and other particulates deposited in receiving waters damage aquatic ecosystems, degrade the aesthetic and recreational value of surface waters, increase flooding and decrease the storage capacity of reservoirs (Novotny & Olem 1994). Nutrients and other chemical pollutants adsorbed to sediments can exacerbate sediments' more direct negative water quality impacts (National Research Council 1993). In the study site used for this analysis, decomposition of sediments contributed by agricultural runoff and streambank erosion severely limits available dissolved oxygen for aquatic life (Suppnick 1992). Thus, this research addresses a very real problem that is of widespread concern.

Sediment Management with Riparian Buffer Strips

One management option that has been shown to be effective is the installation of riparian buffer strips (RBS) (Lowrance 1985). RBS consist of a mixture of grasses, forbs, trees and shrubs adjacent to surface waters that is designed and managed to enhance the capacity of riparian ecosystems to capture and store potential pollutants contained in surface runoff and near-surface throughflow before they can be delivered to surface waters (Dillaha 1989, Osborne & Kovacik 1993, Binford & Buchenau 1993).

Economic constraints usually preclude the installation of RBS along every stream, creek and drain, so a real need exists for a system of prioritizing stream reaches where RBS remediation is most needed and most likely to succeed. While considerable effort has been invested in testing alternative RBS designs, the siting issue has received little attention to date. The approach to delineating variable width investigative riparian buffers presented here is the first step in a strategy to prioritize potential RBS sites. High priority sites are those stream-side locations where the greatest quantities of sediment-laden runoff are likely to be intercepted at the lowest cost both in terms of financial resources in increasingly tight fiscal times, and in disruptions to riparian landowners' desired uses of their land (Maas et al. 1985, Aull 1980).

Hydrological Principles

The investigative riparian buffers presented here are delineated using models based on a combination of hydrologic principles and *ad hoc* judgments of the investigators. The principles are, for the most part, well established in the hydrological literature and are briefly summarized here. Where *ad hoc* judgments were made, these are stated explicitly in the methods or results.

Terrain modeling offers a promising new approach to an old problem: obtaining site specific estimates of sediment movement processes. Researchers (e.g., Walling 1983, Novotny & Chesters 1989) have long sought to overcome challenges in linking upland erosion rates throughout a watershed to sediment yield in receiving surface waters. They have found that the amount of sediment that shows up in the stream is usually dwarfed by that calculated as eroding upslope. The amount of sediment delivered to a stream can be conceptualized as the product of runoff volume and suspended sediment concentration, but both factors are too heterogenous (both spatially and temporally) to make calculations practical. Disturbance of vegetative cover by human activities (e.g., agriculture, timber harvest, and construction) leave soils vulnerable to erosion, but topography, soil hydrologic properties and landscape position largely determine where and how much eroded sediment is actually transported to surface waters via stormwater flows. Eroded sediments typically travel short distances with runoff. Infiltration and declines in velocity which occur when runoff traverses unsaturated soils, surface depressions, low gradient slopes or thick vegetation and when rainfall abates result in deposition of much, if not all, of the sediment load.

Within any given span of time, only a small fraction of sediment eroded within a watershed shows up as sediment in downslope streams. This fraction, formally known as the sediment delivery ratio (Novotny & Chesters 1989), is calculated as the ratio of sediment delivered at the catchment outlet to gross erosion within the watershed. Though the ratio may be a useful conceptual device in discussions of non-point source pollution, it fails to capture the spatially distributed nature of overland sediment transport capacity, thereby conferring a lopsided importance to areas of high hillslope erosion over sources of high water pollution potential. In fact, highly erodible areas far from the stream may contribute far less pollution than less erodible areas close to the stream (Novotny & Chesters 1989).

The difficulty and expense of directly sampling storm runoff has long hampered watershed-scale sedimentation research (Walling 1983). Indirect sampling based on tracking the Cesium^{137} that was deposited on all soils by radioactive fallout during the middle of this century has been used effectively in field studies of sediment redistribution (Ritchie & McHenry 1990). Such studies found evidence that the vast majority of eroded particles from upland fields are deposited in upland depressions, grassed field borders, fence lines, hedgerows and roadside ditches, and for all practical purposes, immobilized indefinitely (Wilkin and Hebel 1982). The research also highlights the importance of riparian condition in determining sediment delivery to streams. For example, replacement of riparian forests with row crop agricultural both places a potential sediment source close to the stream and severely limits the riparian zone's ability to function as a sediment trap for upland runoff (Cooper et al. 1987, Lowrance et al. 1988).

An assumption motivating this work is that water quality modeling and management efforts could be made more effective by re-focusing analysis efforts from all source areas of sediment generation in the watershed to those source areas with a high probability of sediment transport to streams via overland flow. The first step in such an approach is delineation of contributing areas for stormwater discharge to streams because tainted water in such areas has the greatest probability of reaching the streams. The partial or variable source area concept of overland flow suggests that most overland flow in humid environments occurs on portions of the landscape where rain falls on saturated soils, generating saturation runoff. The areal zone of soil saturation expands in the presence of precipitation and contracts in its absence (Betson & Marius 1969, Dunne & Black 1970, Dunne 1978; previous chapters?). Maidment (1993) makes the case that partial flow areas contributing stormwater discharge to streams are predominantly located in the riparian zone:

> "...in most rainfalls on most watersheds, overland flow is not occurring at all, or if it is occurring, the area contributing to runoff is concentrated around the stream network. This concept of localized flow is called partial area flow, and although the concept has been known for about 25 years, it has not been incorporated into many surface water hydrology models because of a lack of ability to determine the size and location of the expanding contributing flow areas as rainfall increases and decreases. By using GIS coupled to models of soil saturation it may be possible to develop more realistic models of streamflow generation...". (Maidment, 1993:p.163)

Soil water redistribution between and during precipitation events accounts for the concentration of surface runoff contributing areas in near-stream positions. Soil water moves both vertically and horizontally, with lateral movement running parallel to surface gradients (Freeze & Cherry 1979, Dunne 1978). Thus, soil water travels downslope following surface gradients and accumulates in flat areas and depressions where low hydraulic gradients inhibit drainage. Relative to the rest of a watershed, riparian zones have three characteristics which make them likely to receive soil water at rates that exceed their drainage capacity: 1) greater upslope contributing areas which cause them to receive more lateral subsurface flow than positions closer to the drainage divide; 2) groundwater table levels that are relatively close to the surface; and 3) relatively low gradients. During precipitation events, the areal zone of soil saturation in riparian areas expands upslope in a non-uniform fashion, with more rapid advances upslope in areas of topographic convergence (i.e., water gathering locations) than in areas consisting of water dispersing ridge lines or uniform slopes.

By itself, soil saturation is a necessary but not sufficient condition for identifying areas with a high probability of sediment contribution. For example, while riparian wetlands have high soil saturation, they also have high capacity to trap sediment as stormwater runoff slowly filters through such low-gradient areas. Delivery of sediments and other NPS pollutants to surface waters via overland flow involves a two phase process: detachment and transport. The velocity of stormwater flow varies with the terrain as it moves overland towards the stream, and where velocity is low, partial or complete redeposition of entrained sediment and other pollutants may occur before they can be delivered to receiving waters (Chesters & Schierow, 1985). The saturated condition of variable source areas facilitates sediment detachment but sediments will only be moved downslope if surface runoff has sufficient and sustained streampower to transport the sediment load. Because of their proximity to receiving waters, pollutants detached in variable source areas in and adjacent to riparian zones have a relatively high land to surface water delivery ratio.

Moore et. al. (1988) modeled the distribution of ephemeral gullies on a 7.5 hectare, bare-fallow cultivated catchment using these concepts. They found that ephemeral gully locations could be predicted from the magnitudes of the compound topographic variables wetness ($\ln(A_b/S)$, where A_b is the local upslope contributing area per unit width of contour line and S is the local slope at the downslope contributing area contour segment) and a stream power index ($A_b * S$). Wetness and stream power were both greatest at the catchment outlet, and high values of both extended some distance upslope along the "valley floors" of water gathering (i.e., convergent) topography. Although not focused on the riparian zone, the strong relationships they found between erosion activity and both wetness and stream power have clear applicability in the riparian zone.

The topographic indices $\ln(A_b/S)$ and $A_b \times S$ assume "steady state" conditions, and are referred to here as static wetness and stream power. Calculation of these indices relies on the assumption that every point in the catchment has reached subsurface drainage equilibrium, and upslope contributing areas can be quite large. Barling et al. (1994) noted some serious limitations of these static topographic indices for describing dynamic flow processes and predicting specific saturated soil and high stream power locations. Most catchment locations only receive contributions of subsurface flow from a small proportion of their total upslope area. This proportion depends upon antecedent moisture conditions and steadily increases throughout the duration of a storm event. Spatially explicit predictions of the distribution of soil moisture and runoff stream power within a watershed requires the ability to describe this "dynamic" upslope contributing area topographic variable.

To this end, Barling developed a quasi-dynamic wetness index that improves on the steady state wetness index by incorporating both the topography within the upslope contributing area and the time required for subsurface drainage to redistribute soil water. Field observations on the same experimental catchment used in Moore's (1988) study verified that the quasi-dynamic wetness index better predicted soil moisture distributions and location of ephemeral gullies than did static indices (Barling et al. 1994). This study represents a logical extension of Moore’s and Barling’s work in an application to riparian management for water quality.

Study Area

Sycamore Creek, an unusually well-studied watershed adjacent to the Michigan State University campus, serves as the laboratory for our riparian vegetated buffer strip siting research. Topography in this primarily rural watershed is flat to gently rolling (Figure 2), and land cover consists (in descending order by area) of a mix of row crop & pasture based agriculture, forest, residential and urban. In 1990, this 27 thousand hecatre watershed centered around the city of Mason, Michigan (population 10,000), and bordered on the north by Lansing (population

100,000), was selected as one of 37 hydrologic units across the U.S. to be included in a United States Department of Agriculture (USDA) and Environmental Protection Agency (EPA) funded demonstration project to provide feedback on non-point source pollution remediation efforts for small watersheds (SCS et al., 1990). The most serious pollution problem identified in Sycamore Creek is stream sedimentation with associated sediment oxygen demand and habitat impairment. Stream bank and agricultural erosion, as well as urban runoff, have been identified as the primary causes of sedimentation (Suppnick, 1992). Sycamore Creek is especially well suited to buffer strip siting research because of its proximity to Michigan State University (which facilitates data collection during storm events), its familiarity to a diverse group of University scientists and agency staff, the community's awareness of and desire to participate in water quality protection efforts, and the availability of relatively complete, current databases. Michigan State University's Institute of Water Research/Center for Remote Sensing (IWR/CRS), the USDA Natural Resource Conservation Service (NRCS), the Michigan Department of Natural Resources (MDNR) and the Ingham County Health Department have assembled a rich data base of environmental descriptors as well as data collected via surface and groundwater monitoring programs. While the time, effort and money expended to assemble information sources that range from multi-spectral satellite imagery to deep well monitoring stations can not be easily duplicated for other watersheds, the rich Sycamore Creek data base can be used to verify more parsimonious diagnostic techniques with broader application potential.

Methods

In the interest of limiting the spatial extent of this analysis (and the concomitant data requirements) during the testing and refinement phases of model development, we narrowed the scope to a single sub-watershed of Sycamore Creek, known as Barnard Drain. After selecting the area of study, four principle tasks remained: acquisition of GIS databases and construction of a digital elevation model of the area, computation of terrain indices via TAPES, formulation and computation of alternative investigative buffer models, and collection and analysis of validation data.

GIS Database and Terrain Model Creation

Our maps of investigative riparian buffers were constructed from three primary GIS data layers obtained for the Barnard Drain subwatershed of Sycamore Creek: streams and drains, topographic contours (10 foot interval), and soils. A GIS coverage (with a base scale of 1:24,000) containing stream and drain center lines for all named streams and drains in Sycamore Creek was obtained from the Michigan Department of Natural Resources (MDNR). The demarcation between streams and drains in this watershed is not distinct- it is not uncommon for streams to be artificially extended upslope as drains, and the bio-physical characteristics of some drains are all but indistinguishable from those of streams of comparable order. Contour data (hypsography) were purchased from USGS as digital line graph (DLG) files on 9 track tape. Contour lines for the southern third of the Barnard sub-watershed were digitized manually from USGS topographic quadrangle sheets because DLG files for those quads were not yet available. Soil polygons and profile attribute data, originally digitized by USDA NRCS were contributed by NRCS for this analysis.

The streams and drains coverage was converted to raster format in preparation for modeling, however, hypsography and soils required significant additional processing.

Topographic sinks were coded in the contour coverage by identifying closed contour depressions on the hypsography layer. A digital elevation model (DEM) for the Barnard subwatershed was constructed from the hypsography theme using ARC/INFO's TOPOGRID module (see Hutchinson, this volume). TOPOGRID was executed using recommended tolerances with drainage enforcement to the streams and drains layer and the sinks file to create output with a ten meter grid cell size. While the resulting DEM surface appeared to be relatively smooth, histograms of cell frequency by elevation value demonstrated the distinctly stair-step signature that has been widely observed by those who have relied on contour interpolation in regions of gentle topographic relief (e.g., Hutchinson chapter in this book; Eklundh and Martenson 1995). Contrary to our expectation that this artifact of the interpolation algorithm would lead to locally elevated wetness index values (due to relatively lower slope values in the vicinity of contour lines), no such phenomenon was observed. We therefore accepted and used the results from TOPOGRID as an adequate representation of the topography in the area.

Although TAPESG does not rely on soil properties to calculate static wetness and stream power, DYNWETG does use saturated hydraulic conductivity (K) and effective porosity (P) information along with slope (S) in calculating dynamic indices via a formula for average lineal velocity (VEL) of soil drainage (i.e. the speed of near-surface throughflow) based on Darcy's law:

$$VEL = KS/P$$

In this calculation, percent slope at the surface, easily derived from a DEM and automatically included in TAPESG output, serves as a surrogate for hydraulic gradient. Field measurements of K and P, however, are extremely challenging to obtain and these variables are usually quite spatially heterogeneous. DYNWETG accepts K and P as spatially uniform parameters (averages over the study area) or as spatially variable parameters in the form of raster files. Both uniform and variable soil parameter assumptions were included in this analysis. Uniform averages and variable raster files of K & P were estimated from surrogate measures in the USDA NRCS Ingham County Soil Survey digital database associated with the digitized version of the county soil survey map. Permeability, measured in a laboratory environment as the rate of vertical movement of water through a soil column in inches/hour, served as a surrogate for K. The mean of permeability (weighted by horizon thickness) for all surface soil horizons provided the single value per soil type required by DYNWETG. Because lateral movement of soil water is dominated by near-surface flows, only surface soil horizons were considered. We identified surface horizons as those both above the recorded high water table level and above the first aquiclude (defined here as a fine textured soil horizon with permeability <= .005" / hour).

Drainable porosity, estimated from soil texture class (e.g. loamy sand, sandy loam, loam etc.) and a table relating texture classifications to drainable porosity (Foth 1984: Figure 3-12), was used as a surrogate for effective porosity in the models described in this chapter. Alternatively, a table which permits a direct lookup of effective porosity based on soil texture is given by Rawls et al. (1992), and those seeking to emulate or extend this terrain analysis application are advised to consult it.

Generation of Terrain Analysis Indices

Our analysis begins with the use of the simplest models of wetness and stream power. Then we modify the assumptions to test the sensitivity of the results to more elaborate models. The Barnard DEM was processed twice with TAPESG: once using the D8 flow-routing algorithm, and once using DEMON. For each TAPESG run, the outputs slope (S), flow direction, and upslope contributing area (A) were used to calculate static indices and to provide inputs to DYNWETG. Alternatives to the D8 flow-routing algorithm are not currently available in the DYNWETG software. We produced quasi-dynamic indices using both uniform and distributed soil parameter assumptions with a 24 hour drainage time. The distributed soil parameters were produced as described above. The principle DYNWETG output used in our models was Effective Upslope Contributing Area (A_e).

A total of four sets of wetness and stream power indices, beginning with the simplest, were ultimately generated from upslope contributing area (A), effective upslope contributing area (A_e), and slope (S) to serve as the basis for four models: 1) Static D8, 2) Static DEMON, 3) Dynamic Uniform Soils, and 4) Dynamic Variable Soils. For the static cases, topographic wetness index (TWI) was calculated as TWI = ln (A/S) and stream power (PWR) as PWR = AS, and for the dynamic cases, TWI = ln (A_e/S) and PWR = A_e S.

The contributing area and slope components were transferred to Arc/Info grid files using a conversion routine in the TAPES package (TAPESTOARC) and ESRI's ASCIIGRID command. Wetness and stream power indices were calculated using Arc/Info GRID from the terrain primitives; however, one could just as easily obtain these from the output files produced by TAPESG and/or DYNWETG. One caution is that, in order to produce absolute and comparable results, keeping track of units is essential (e.g., is slope in degrees, percent, or as rise over run; was TAPESG run for areal units or numbers of cells) as they can have great impact on the magnitude of calculated index values.

Formulation of Investigative Buffer Model

Building on Moore et al.'s (1988) conclusion that indices of both wetness and stream power provided predictive power in modeling the locations of ephemeral gullies, with relative predictive power of these indices determined by slope position, we sought to build a model that would combine these indices to identify likely storm water contributing areas. Our approach is similar to Moore's, in that we are combining wetness and stream power maps to find areas with high levels of both, but differs in that we identify areas of high wetness and high stream power as fuzzy, rather than boolean, sets (Burrough, 1989). We used fuzzy logic to create continuous surfaces for indentifying contributing areas because of highly skewed distributions of stream power compared with fairly normal wetness distributions. The fuzzy logical approach was used to assign locations to sets called "high stream power" and "high wetness" in a watershed-relative fashion based on percentiles within the distribution (described below).

The fuzzy intersection of the wetness and stream power sets served as input to the calculation of cost distance from the stream channel, which was used to identify variable buffer widths and focus the investigation on near-stream locations. Presumably, areas with high values of both stream power and wetness merit investigation for possible sediment contribution. Because continuity between sediment production and the stream is required for NPS pollution to occur, we used the cumulative cost distance calculation from the stream to construct a buffer which was wider where sites with high stream power and wetness occurred near the stream and narrower where values of stream power or wetness were lower near the stream. The fuzzy intersection of wetness and stream power, then, served as a unit cost input to cumulative cost distance calculation, where locations with higher values of wetness and stream power correspond with lower unit cost so that cumulative cost distance contours bulge out from the stream at such locations.

We created two fuzzy membership surfaces for each different model, one for wetness and one for stream power. The fuzzy sets on 0.01 - 1.00 intervals were designed so that cells with the greatest values of wetness and stream power would be assigned the minimum cost and those with the lowest values (and thus least likely to coincide with detachment and transport) would be assigned the maximum cost. The fuzzy membership functions for each index were defined such that they decline linearly from 1.00 to 0.01 from the 50th to the 95th percentile on the stream power and wetness distributions, as follows:

Percentile range	Fuzzy set value
0-50	1.00
50-95	$1-0.99\left[\frac{index-50^{th}Percentile}{95^{th}Percentile-50^{th}Percentile}\right]$
>95	0.01

The intersection of two fuzzy sets (i.e., the assignment of a value of membership in the set formed by the intersection of Set A and Set B) can be either hard, using the MINIMUM operation, or soft, using multiplication (Burrough, 1989). We used the soft intersection operation to assign unit cost distance values. Resultant values were low where wetness and stream power were both high relative to the rest of the watershed.

Cumulative cost distance values, basically the minimum sum of unit cost distances traversed between a cell and the nearest (cost distance) stream cell, were assigned to each cell by accumulating the cost values as recorded in the fuzzy intersection map starting from the stream. From the cumulative cost distance map, a map of the investigative riparian buffer was generated by assigning the lowest n percent of values to 1 and all others to 0. The selection of n was dependent on watershed specific criteria and could be sliced using multiple threshold percentiles to obtain bands of progressively diminished concern (e.g., 0-5%ile, 5-10%ile, etc.). Note that increases in n do not expand the buffer at a uniform rate or in a self-similar fashion across every segment of stream.

Collection of Validation Data

Two types of validation were performed for these models. Because the traditional approach to investigative riparian buffer siting is based on map interpretation by water quality experts or field observations of riparian condition, these alternatives make appropriate benchmarks against which model performance can be judged.

Expert Interpretation of Topographic Maps

Two experts from the hydrologic unit at the Michigan Department of Environmental Quality (MDEQ) Land and Water Management Division were provided with 1:24000 scale USGS topographic maps covering Barnard Drain and asked to identify flow paths where they would expect to find surface flow destined for the stream, based only on the clues provided by topography. In essence, they were asked to manually interpret the maps to find coincidence of convergent topography and evidence of sufficient steam power to give surface flow a high probability of reaching the stream. The correspondence between these experts' assessment of the routes that surface flow would follow (i.e., linear features) and the predictions of the wetness indices can be checked to provide a form of validation of either approach.

Delineation of sub-watersheds at this kind of scale for engineering purposes and fisheries and water quality assessment and monitoring is a daily activity for experts at MDEQ's hydrologic unit. These experts had some difficulty complying with our request because they were accustomed to operating with some design points in mind (e.g., specifying culvert dimensions based on predicted flows) and delineating upslope area on maps to identify sub-watersheds; pre-delineating a watershed before a problem has been identified which would provide them a context was clearly a novel undertaking for them. It is also worth noting that they repeatedly asked to be allowed to use additional information, such as soil survey data, and aerial photo stereo pairs, in making their judgments, probably in search of clues concerning land cover and topography. The flow paths drawn by the MDEQ staff were digitized to create a GIS coverage and compared visually with output from the models.

GPS Referenced Field Observations of Wetness Indicators

If the technique of using terrain analysis wetness indices to predict surface flow is to prove fruitful, it is surely reasonable to expect greater values of wetness index where saturated conditions are observable in the field. Stream power might or might not be greater at such locations, depending principally on slope. Presumably, wet areas close to the stream have a greater likelihood of contributing sediment detached locally, and of transporting to the stream sediment generated farther upslope. To test the correspondence between observed wetness and the terrain based indices, we surveyed the full length of both sides of Barnard Drain over a period of a few days in April, 1995, shortly following a series of significant rain events, to identify sites with indications of soil saturation or erosion activity. Using an integrated GPS/data logger, 24 sites showing evidence of wetness were geo-referenced as points and attributed as wet spots (soil saturated and/or ponding evident), drain points (evidence of concentrated flow but vegetation still present), and small gullies/bank erosion sites (vegetation and/or topsoil apparently removed by the force of concentrated flow). Relative wetness and stream power for each of these 24 points, which we refer to generically as "wet spots", were calculated to determine whether our expectations would be realized in Barnard Drain.

Wet spots were converted to raster grid cells, then expanded from one grid cell (10 m by 10 m) to areas that were three by three grid cells (i.e., 30 m by 30 m) in size using GRID's FOCALMAX operator. This "fuzziness" was added because 1) even with differential correction, GPS based location estimates can easily be off by one cell in a 10 m grid and 2) the tendency of D8 flow routing to constrain flow paths to even multiples of 45 degree azimuths could produce artifacts which introduce positional errors of as much as one cell. Because most of the recorded wet spots were within a cell or two of the stream, these three by three grid cell areas tended to overlap the stream or, in some cases, portions of the opposite bank. Cells in a rasterized version of the USGS blue-line representation of the stream and cells on the opposite side of the stream were removed from the areal representations of wet spots, leaving a grid of wet patches with a variety of shapes and sizes (between two and nine grid cells).

Stream power and wetness for all grid cells in each wet patch were averaged and compared with the distribution of values within a 30 meter buffer around the stream to obtain relative (percentile) representations of these variables. Thirty meters was chosen because it was the minimum buffer distance that included all but one of the wet spots. The statistical distributions of values of stream power and wetness, derived using each of the various methods described above, were summarized using percentiles (i.e., percentage of values below a given value) calculated on all cells within the 30 meter buffer (n = 1666). The percentiles were then calculated for the average of the multiple grid cells in each wet patch. High percentile values (i.e., closer to 100), particularly for wetness, indicated consistency between the model and observations on the ground in that these were among the wettest sites near the stream.

Results

Three types of results were produced for each of the four alternative models: the calculated wetness and stream power indices, the variable width investigative buffers, and degree of agreement with the validation dataset. Outputs among the models are contrasted and some potential explanations explored.

Comparison of Indices

Given the multiple order of magnitude difference between A (mean=601, max=191859), which represents an accumulation of contributing cells all the way to the top of the watershed, and A_e24 (mean=5.1, max=375), which represents only those cells from which water would flow following a 24 hour precipitation event, it is not surprising that maps of stream power and wetness calculated from these measures look entirely different (Figure 3). Both effective and upslope contributing area are highly left skewed (Figure 4). While wetness (both static and dynamic) is logged and can be approximated by a normal distribution, stream power is not. In the static case, regardless of whether flow routing is by DEMON or D8, stream power in the channels is so great (due to the high values of A), that the variable cannot be scaled such that variation is visible elsewhere. Dynamic stream power shows more variation outside of the channel network than does static, demonstrates a distinctly checkered pattern with discernable diagonal trends characteristic of D8, and appears to be controlled most strongly by slope in upland areas, and by contributing area near the channels (Figure 5).

Differences between wetness maps for the dynamic cases (variable and uniform soils) were great and those between the static cases (DEMON and D8 flow routing) were even more pronounced, but both were dwarfed by the differences between static and dynamic cases (Figures 3 and 6). The static wetness model which relied on DEMON flow routing exhibited much smoother spatial transitions between high and low wetness values which may represent an aesthetic improvement, but there is a substantive difference as well: the drop-off in relative wetness value with distance from the stream is more gradual. While every potential channel is clearly demarcated by high values of static wetness, contiguous areas of high dynamic wetness are less common and somewhat less pronounced.

Comparison of Variable Width Investigative Buffers

While it may be possible to identify areas of relatively higher storm water contribution by visually interpreting the cost distance map derived from the cost surface created by the fuzzy intersection of wetness and stream power (Figure 7), most watershed managers would likely prefer a map product delineating an investigative buffer within which attention should be concentrated. One way of creating such a map is to select a percentile threshold for cost distance, below which is buffer, and above which is not. Without further field validation and analysis, it is impossible to arrive at an empirically determined threshold. As a first approximation to facilitate our evaluation of the approach, we made the ad hoc decision to select the fifteenth percentile as the threshold for the example buffers shown in the following figures.

All four models generated what could be charitably described as variable width investigative buffers; however, there were striking differences in form among them, and those generated by the static models bore a closer resemblance to a dendritic network than the kind of variable width investigative buffer that we had in mind when we undertook the analysis. The buffers created using static wetness with both the D8 and DEMON flow-routing algorithms were quite similar to one another (Figures 8a and 8b). The primary observable difference was the presence of numerous strands of buffer oriented in an even multiple of 45 degrees, undoubtedly an artifact of the D8 requirement that all flow be routed to a single cell, and the relatively even topography characterizing this watershed.

The dynamic models produced buffers that were much more consolidated than the static models, and which more clearly flagged segments of the stream meriting closer investigation to assess existing sediment filtration capacity and opportunities for remediation. The uniform and variable soils dynamic models produced buffers that were quite different from one another, with the uniform soils buffer tending towards the dendritic structure dominant in the static models (Figures 8a-d).

The choice of threshold for cost-distance has a significant impact on buffer delineation, and buffer thickness does not expand in any kind of predictable fashion as the percentile threshold increases. A 15 %ile threshold generates a buffer on far more segments of stream than does a 5%ile buffer (Figure 9). We chose the 5% buffer for all subsequent figures to limit investigative buffer size and focus only on the areas most susceptible to NPS pollution.

Concordance with Validation Dataset

Flowlines

Overlay analysis of digitized flow paths on cost buffers produced by the four models revealed that buffer strands coincided with all or portions of the flow paths for all 16 of the flow paths for both static models, for 6 of the flow paths in the variable soils dynamic model, and for 12 of the flow paths in the uniform soils dynamic model. However, for the static models, there were many "false" buffer strands (at least if the flow paths delineated by MDNR staff are considered ground truth), and overall, about a 2:1 ratio of false to true buffer strands (Figure 8).

The reasons for the failure of the dynamic models to generate wider buffers or buffer strands for some of the flow paths were evident when flow paths were overlaid on dynamic wetness index (Figure 10). In several cases (4 of 16), it appeared that modeled wetness was low because what might appear to be a flow path encountered a flat or water-dispersing area before reaching the stream (Figure 11a). Because neither dynamic wetness nor stream power maintained high values all the way to the stream, and because the buffers are constructed as a cost distance from the stream, it is not surprising that these models did not generate wide buffers in such instances. Moreover, it is entirely appropriate for these areas not to have wide buffers, as the likelihood of sediment transport to the stream under such conditions is relatively low. In other cases (3 of 16), lack of coincidence could be attributed to differences between the human interpreters and TOPOGRID in locating drainage routes that sometimes led to offsets of as much as 50 meters (Figure 11b).

In contrast to the static models, the dynamic models produced relatively few "false" wide spots and strands. The uniform soils model, for example, produced only a handful of these false positives.

Some of the flow paths that the MDNR interpreters had the most difficulty deciding about involved large sub-catchments with low gradients near the bottom of the catchment. The interpreters were uncertain as to whether they should include such flow paths because they were not confident that such flow would ultimately extend to the stream. These were the same flow paths that were not matched by wide buffers in the dynamic models.

Wet Spots

Results of the validation using field observations point to more agreement than differences among the models. And, for every model, the expected values of both wetness and stream power for the 24 observed wet spots are in the top half of the distribution of such values within 30 meters of the stream. Furthermore, these percentile distributions are all right skewed - dropping even the two or three points with the lowest values (when there is justification for doing so, such as the fact that points 117, 118 and 411 could not be differentially corrected and thus may appear up to 8 cells from their true locations, and quite possibly outside the established 30 m buffer) brings the average index values up considerably. Because all the features in this dataset were associated with wet spots (rather than high transport potential), we expected to find better concordance in the wetness indices than in stream power; however, the statistics appear to contradict this expectation. In every case, mean stream power percentile was greater than mean wetness percentile.

Although the D8 Static model generated the largest mean wetness percentile (72), the Variable Soils Dynamic model was a close second at 70, and is preferred over the others for the reasons related to the flow line validation and investigative buffer form outlined above. An improved correspondence might be established if observed wet areas are digitized as polygon rather than point data.

Table R1. Relative index values for field observed wet spots for each model as percentile of all cells within a 30 meter buffer about the stream. Rows are sorted in descending order by dynamic wetness index for the variable soils case.

			Percentiles							
			D8		DEMON		Uniform		Variable	
Point#	Point Type	Cells	Pwr	Wet	Pwr	Wet	Pwr	Wet	Pwr	Wet
408	BANKEROS	3	88	88	64	66	95	96	100	100
107	WETSPOT	3	90	92	55	62	92	98	92	99
124	DRAINPOINT	3	98	90	87	23	99	94	100	99
403	DRAINPOINT	3	89	88	95	86	88	86	99	99
112	WETSPOT	3	90	94	35	30	90	97	88	98
237	DRAINPOINT	3	91	87	94	88	93	87	97	97
505	DRAINPOINT	3	54	53	56	70	56	56	94	97
410	DRAINPOINT	6	88	85	28	82	78	86	87	95
218	DRAINPOINT	6	90	85	89	55	98	77	100	93
108	WETSPOT	3	89	96	88	91	28	88	18	91
233	DRAINPOINT	4	88	88	32	81	44	89	35	91
216	DRAINPOINT	6	93	82	82	16	99	65	100	78
220	DRAINPOINT	6	90	83	23	11	98	60	100	71
411	DRAINPOINT	3	86	87	93	90	55	93	6	69
405	BANKEROS	2	83	83	90	94	33	84	13	67
214	DRAINPOINT	5	92	74	43	4	99	43	100	62
222	DRAINPOINT	9	40	67	43	72	35	61	36	59
117	WETSPOT	9	72	42	69	35	84	47	92	50
122	DRAINPOINT	4	79	28	86	34	87	28	96	48
207	BANKEROS	4	80	20	65	7	82	14	93	35
118	DRAINPOINT	9	27	24	70	59	39	27	56	33
212	DRAINPOINT	5	96	84	76	6	98	43	98	21
228	DRAINPOINT	3	67	55	60	61	52	47	21	17
512	BANKEROS	2	74	60	81	70	80	62	26	11
MEAN			81	72	67	54	75	68	73	70

Discussion

The considerable differences in form and pattern among buffers produced using different modeling assumptions demonstrates that buffer delineation is highly sensitive to the choice of wetness index and soil parameters (in the case of dynamic wetness), and less sensitive to the choice of flow routing algorithm (in the case of static wetness). This last finding is encouraging, as the software does not yet exist to implement the DEMON routing algorithm into dynamic wetness index calculations. As noted in Chapter ?? of this book and elsewhere (e.g., Costa-Cabral and Burgess, 1994), the DEMON algorithm provides a more realistic representation of two dimensional flow and produces more accurate estimates of specific contributing areas. Yet, if the results of the static model comparison are at all valid, a DEMON based dynamic wetness based buffer could be expected to be somewhat smoother and less fragmented than those produced by D8, but not have an appreciably different pattern of buffer thickness along the stream.

Of the models tested, the variable soils dynamic wetness model produced buffers that most resembled the idealized variable width investigative riparian buffer envisioned at the outset, and that stand the best chance of being interpretable by watershed managers seeking to prioritize sites for riparian buffer strip installation. This model also ranked second in correspondence between field identified wet spots and high wetness index values.

Transport appeared to be far better represented by the dynamic model, in that buffer widths reflected terrain induced reductions in stream power and wetness present along potential flow paths. Because static models presuppose flow from the entire upslope contributing area (extending to the "top" of the watershed), nearly every potential flow path is represented as a contiguous chain of cells with high wetness and stream power. This surely represents a departure from all but the most infrequent (e.g., 100 year storm) precipitation events.

The variable soils dynamic model best matched the flow path delineations of MDNR experts, when "false positives" are taken into account. The general agreement between modeled wetness (and buffers) and flow paths interpreted from topographic maps should instill confidence in the approach in the minds of watershed managers.

For these reasons, the variable soils, dynamic buffer appears to be the best choice for riparian investigative buffer delineation. However, further work is needed to ascertain the best drainage time settings and cost distance thresholds. Additional modeling must be supplemented with more extensive and rigorous field validation of soil moisture that may well involve dynamic monitoring at a fine spatial resolution to relate saturated conditions to storm events of various durations. Ideally, validation would also entail measurement of site specific water and sediment flow during storm events and spring melt.

Logical extensions of this work include several alternatives for integrating land use/cover information (Phillips 1989, Xiang 1993). An easy extension to implement would be to prioritize existing land uses within the investigative buffer for riparian buffer strip installation. A prerequisite to this extension is evaluation of the natural buffering capacity (e.g., sediment trapping capacity) of different land cover types via field measurements during and following precipitation events. Then, these land types would have to be mapped within the investigative riparian buffer (Figure 12) so that those land uses with little inherent buffering capacity and those deemed most likely to be net contributors of potential pollutants could be efficiently targeted. However, current land use/cover information can be difficult to obtain, and in areas of rapid land use turnover, may need frequent updating (e.g., from recent aerial photographs). A fundamental advantage of the investigative buffer approach is that such information would be needed for an area far smaller than the whole watershed (e.g., the 5-15% of the watershed bounded by the investigative buffer).

A somewhat more difficult extension would address the nature of land cover (and pollution generating capacity) upslope from each cell. The models presented here were designed to identify areas of surface flow extending to the stream; but no accounting for upslope land cover is considered. Thus, a wide buffer with polluted water contributed from an upslope feedlot or agricultural field is indistinguishable from a wide buffer with clean water contributed entirely from an upslope forested area. If weightings or pollutant loadings can be calculated for different land uses, these could be accumulated in the calculation of dynamic indices with some modification of the DYNWET software. However, any gains in explanatory power resulting from the endogenous inclusion of land use in such a model must be weighed against the costs of the additional data needed to implement it.

Finally, to be effectively implemented, such technical modeling efforts must ultimately be integrated with institutional realities and the preferences of riparian landowners. A combination of terrain modeling and land use/cover mapping may be useful for selecting riparian segments meriting a high priority for remediation effort, but some kind of regulatory or incentive structure will likely be needed to translate such analyses into remediation on the ground. Ownership patterns (e.g., parcel size distributions) may also be important in the siting decision if costs of education and technical assistance are high on a per landowner basis. Because the topographic predisposition of riparian segments towards being variable source zones of potential pollution is relatively static (at least on human time scales), counties, townships and drain commissions may find it useful to denote terrain predisposition on property records to facilitate targeting of landowner contacts and assistance efforts.

Conclusions

Terrain modeling of contributing areas for stormwater discharge appears to be an application of terrain analysis with great promise for guiding site specific water pollution remediation efforts. Analysis for large watersheds should not be appreciably more difficult or time consuming than for small ones, so the potential for scale economies exists. While there is general agreement between maps produced by terrain modeling and those produced by human interpretation of topographic maps, the areas of disagreement are even more interesting. The areas of greatest disagreement tend to be relatively flat, lending credence to the conventional wisdom that flow path determination by any method can be especially challenging in areas of low relief.. Preliminary validation of model outputs against field observations of wet spots provided encouraging evidence of model accuracy.

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Figures

Figure 1. (a) Investigation of the whole watershed at a resolution suitable for modeling yields little information depth; (b) re-deployment of the same investigative effort within the hydrologically active riparian zone enables the collection of a richer attribute data set for supporting modeling and planning.

Figure 2. The Barnard Drain sub-watershed: (a) Drainable soil porosity, (b) Saturated hydraulic conductivity, (c) Topography (10 foot contour interval), hydrography and highway

Figure 3. Index maps of stream power and wetness based on static and dynamic uniform soils models. Maps were generated using a linear gray stretch such that lighter shades correspond to high index values.

Figure 4. Histograms of upslope contributing area calculated with (a) static DEMON (A) and (b) dynamic, uniform soils (Ae) models for Barnard Drain. Note the log scales on the x-axes.

Figure 5. (a) Slope, via finite difference method, as computed by TAPESG and (b) Ae (Dynamic, uniform soils) for Barnard Drain.

Figure 6. (a) Static wetness index computed with DEMON flow routing algorithm. (b) Dynamic wetness computed using variable soils and a 24 hour drainage time. Both maps generated using linear gray stretch, with lighter shades representing higher index values.

Figure 7. Cost distance values generated using dynamic wetness and variable soil parameters for a portion of Barnard Drain displayed using a linear gray

stretch with the stream overlaid. Darker shades correspond to the lowest cost distance values (i.e., cells that are close to the stream and have relatively high wetness and/or stream power).

Figure 8. Cost-distance buffers at the 5%-ile threshold with interpreted flow paths (dotted lines) overlaid for the (a) static, D8, (b) static, DEMON, (c) dynamic, uniform soils, and (d) dynamic, variable soils models.

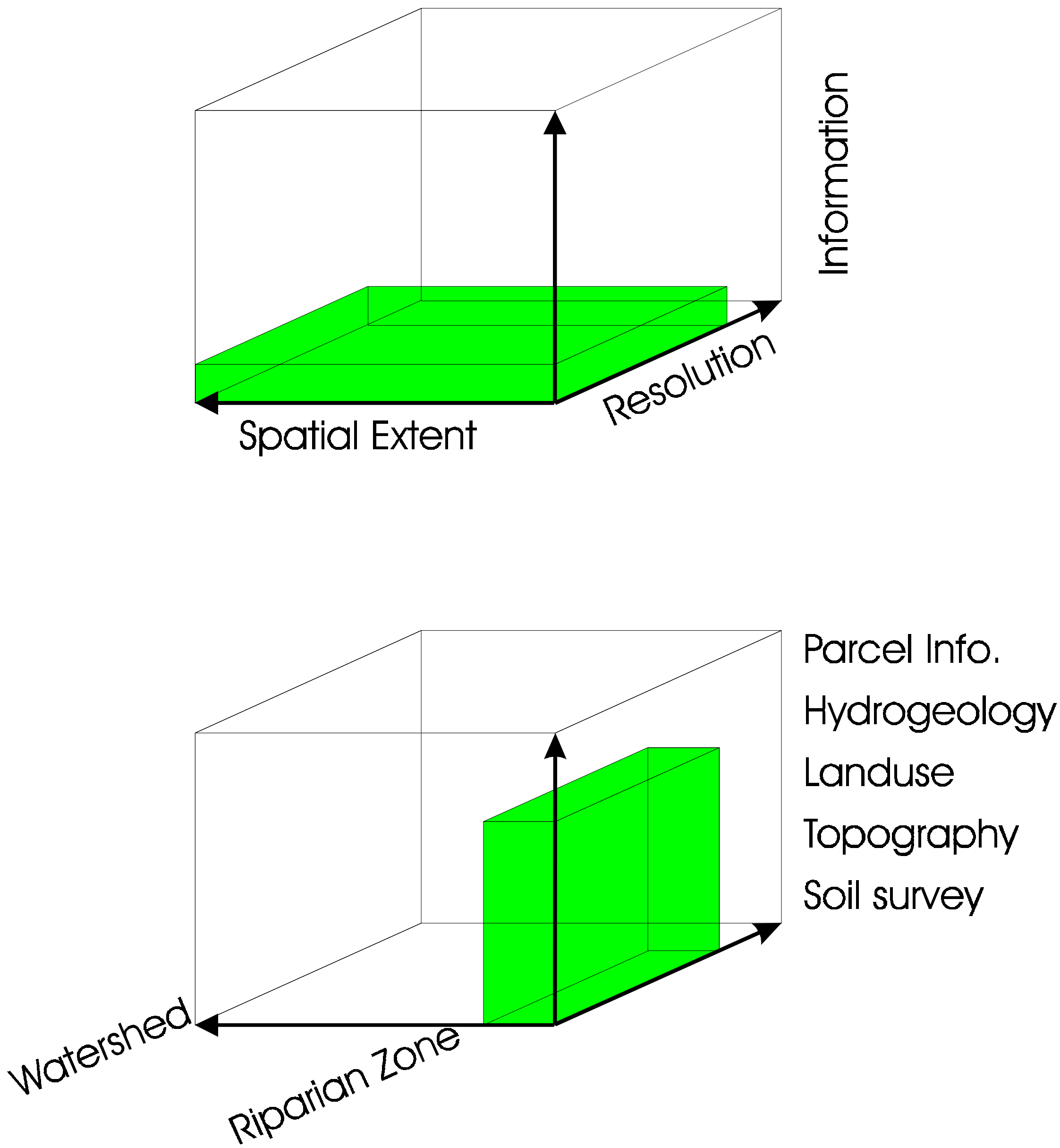
Figure 9. Cost-distance buffers for 5 (black), 10 (black + dark gray), and 15 (black + dark gray + light gray) percentile thresholds generated from the dynamic, variable soils model.

Figure 10. Interpreted flow paths (green) and 10 foot contours (yellow) overlaid on wetness index maps (linear gray stretch) for a portion of Barnard Drain produced by (a) static, D8, (b) static, DEMON, (c) dynamic, uniform soils, and (d) dynamic, variable soils models.

Figure 11. Potential flow paths (strands of high wetness) that (a) disperse due to flat topography and (b) diverge from flow paths manually delineated by watershed experts.

Figure 12. Land use within the investigative buffer at the five percentile threshold produced by the dynamic, variable soils model.

Figure 1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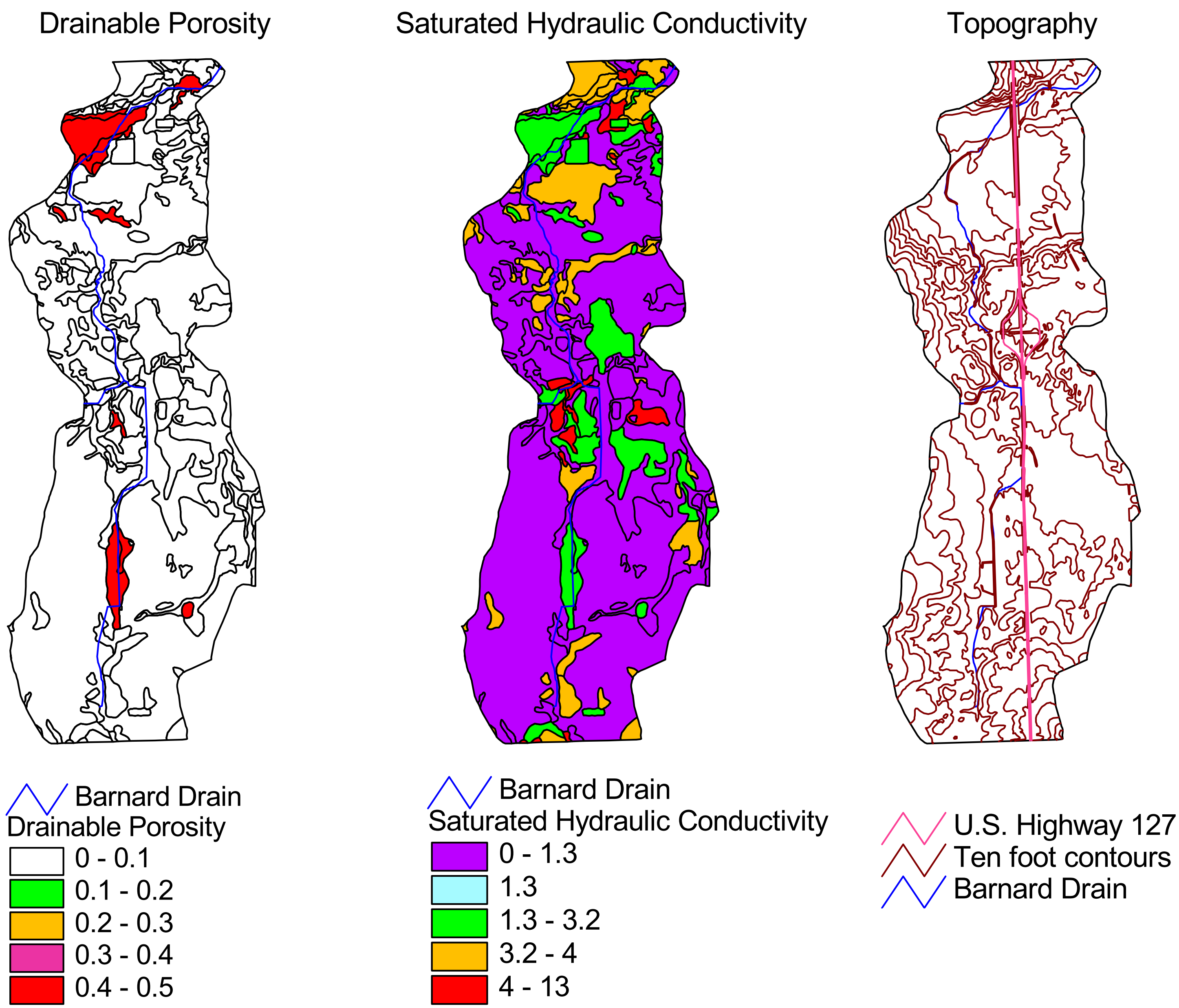
Riparian Zone


Drainable Porosity
Saturated Hydraulic Conductivity
Topography
Barnard Drain
Drainable Porosity
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0.3 - 0.4
0.4 - 0.5
Barnard Drain
Saturated Hydraulic Conductivity
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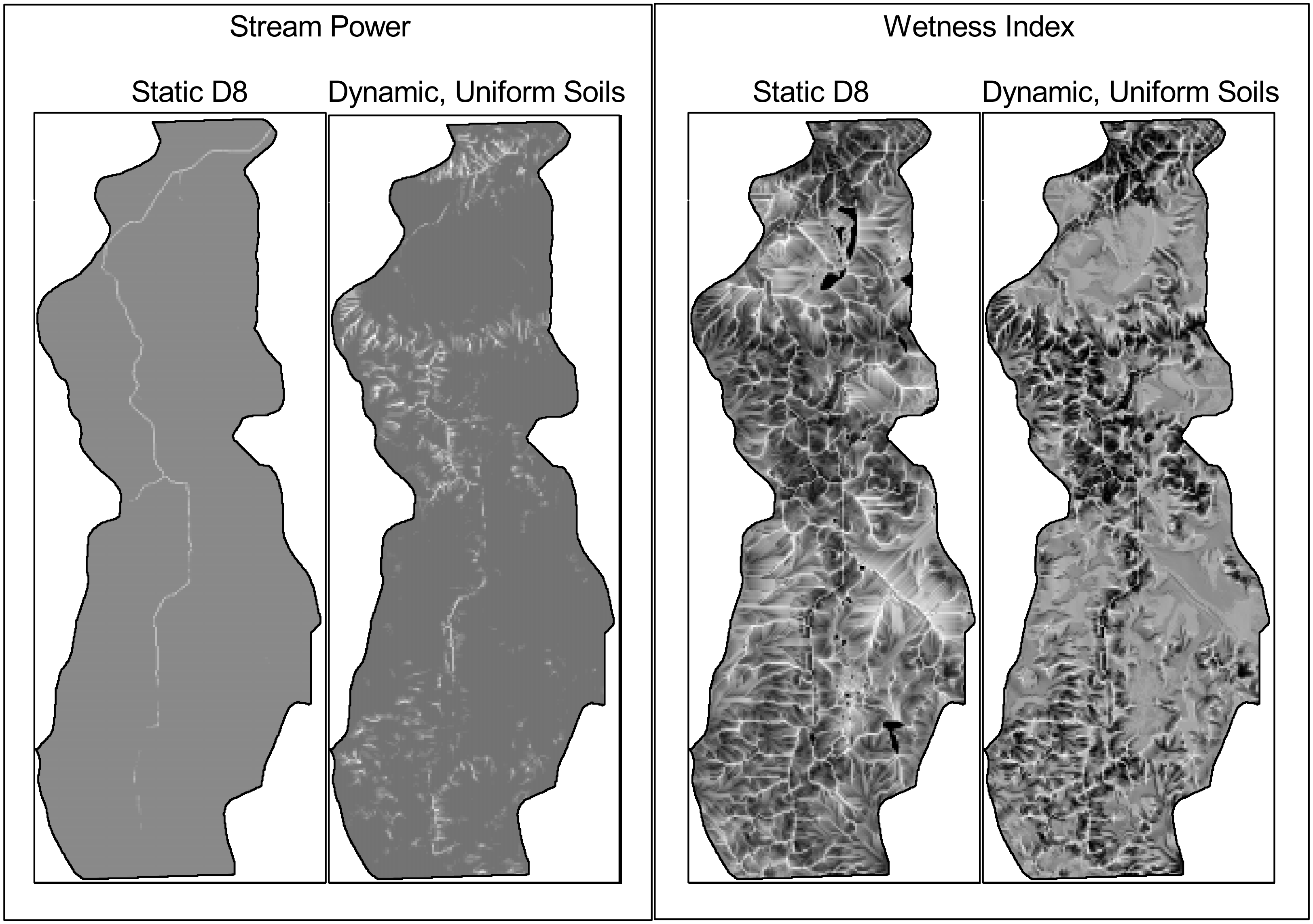
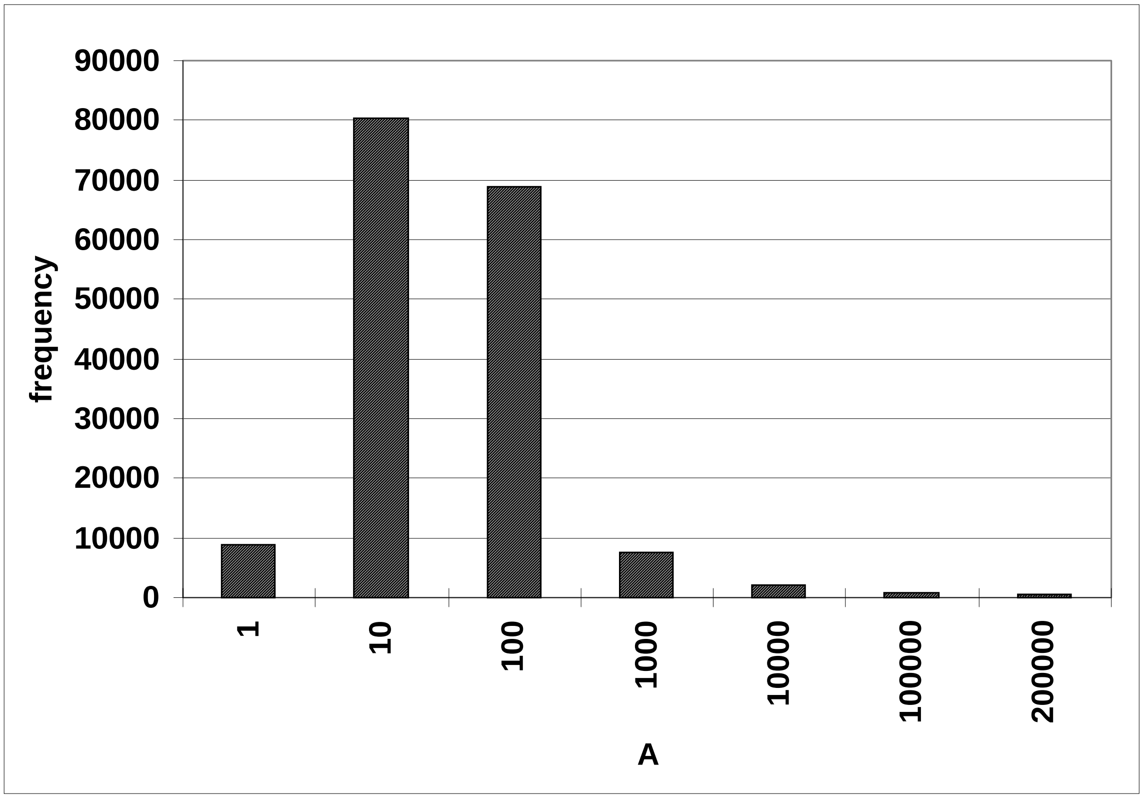
1.3
1.3 - 3.2
3.2 - 4
4 - 13
U.S. Highway 127
Ten foot contours
Barnard Drain

Figure 2.



Stream Power
Static D8
Dynamic, Uniform Soils
Wetness Index
Static D8
Dynamic, Uniform Soils

Figure 3.



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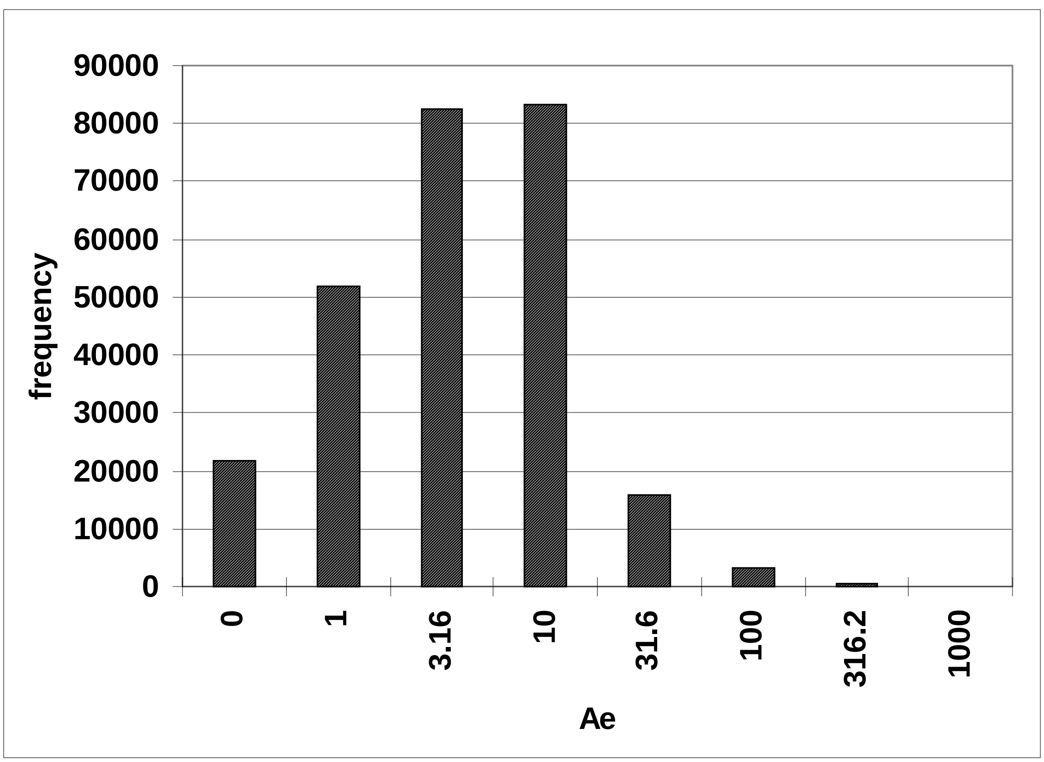

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Figure 4

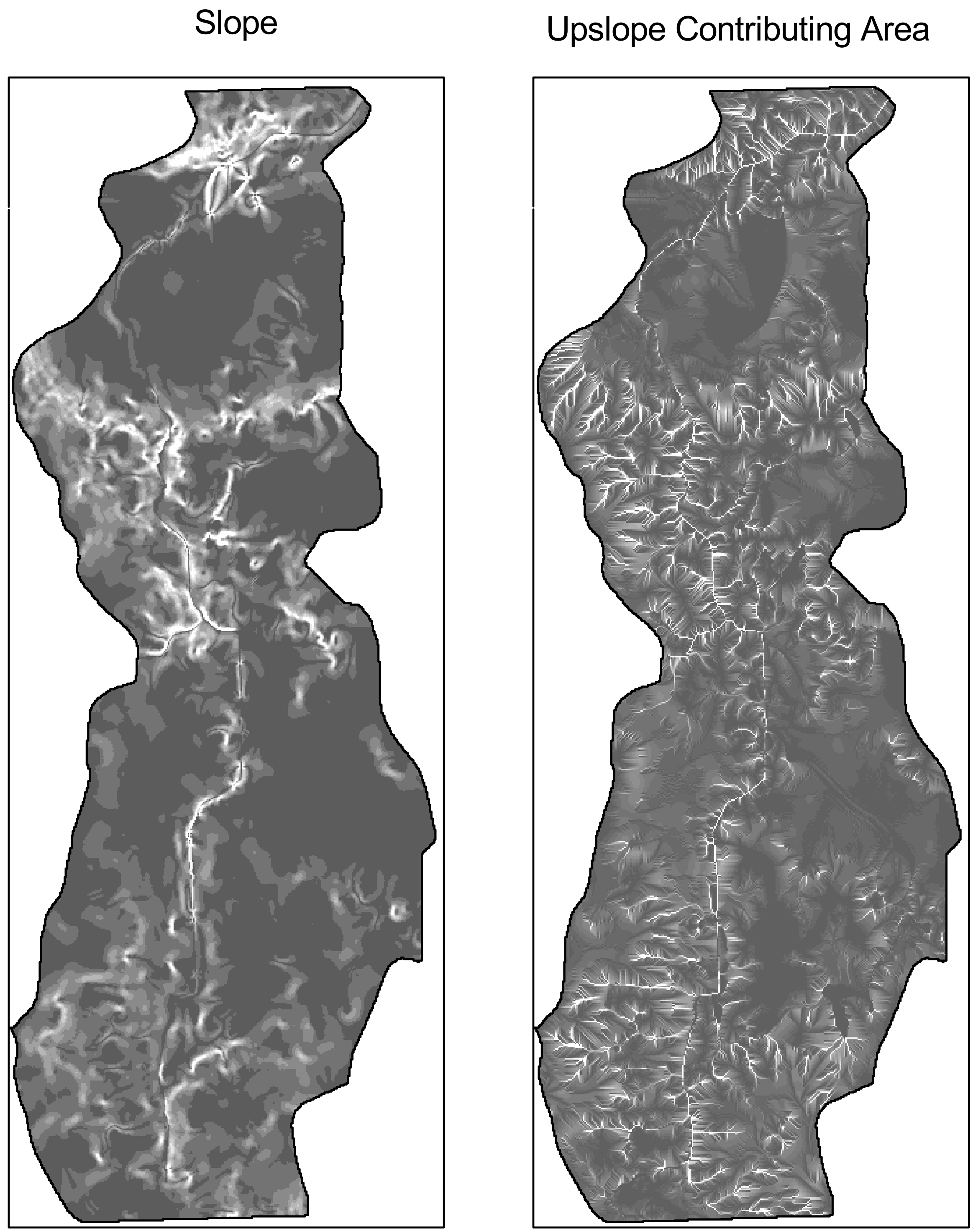


Figure 5.

Static DEMON

Dynamic, Variable Soils

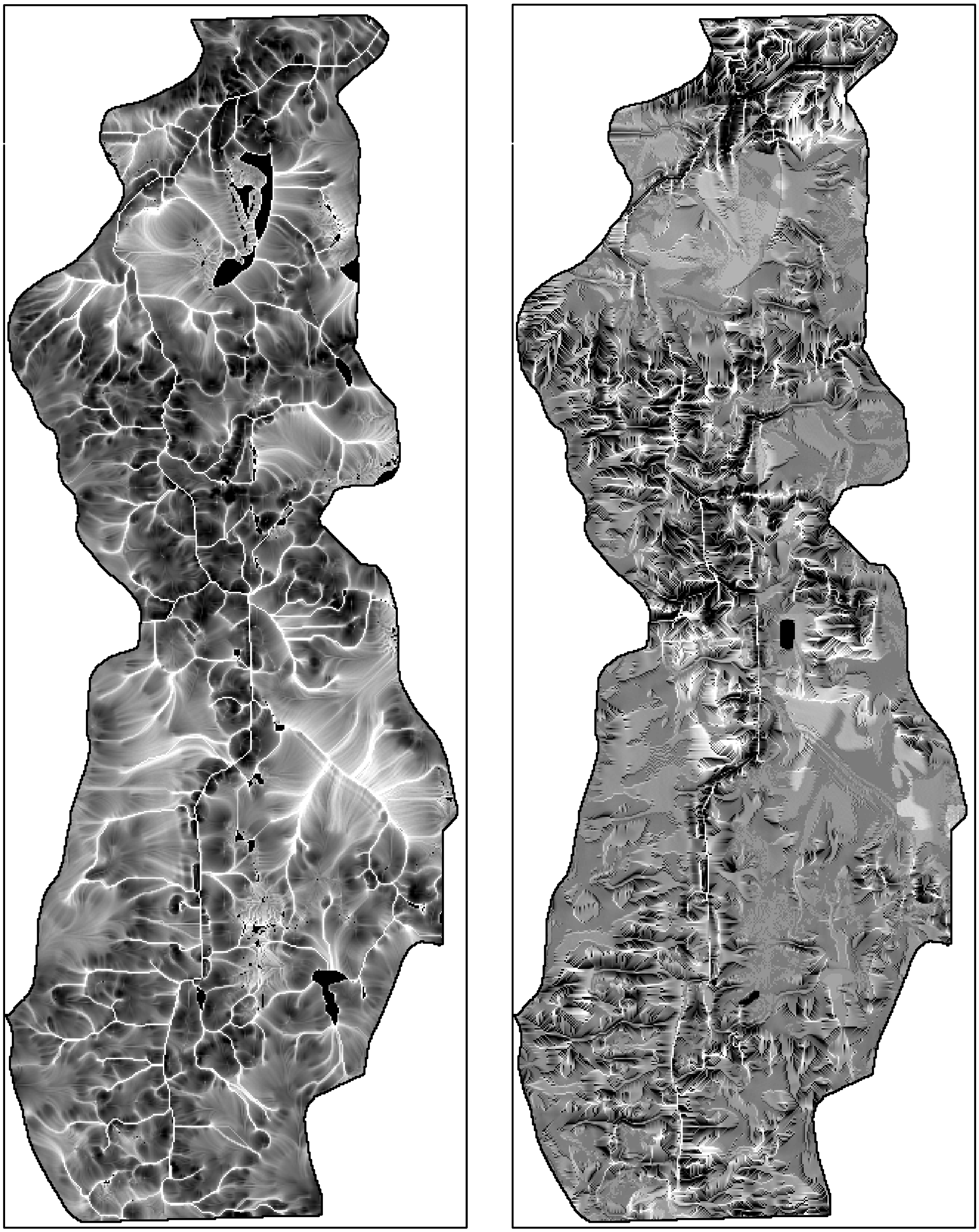


Figure 6.

Dynamic, Variable Soils -- Cost Distance

Figure 7.

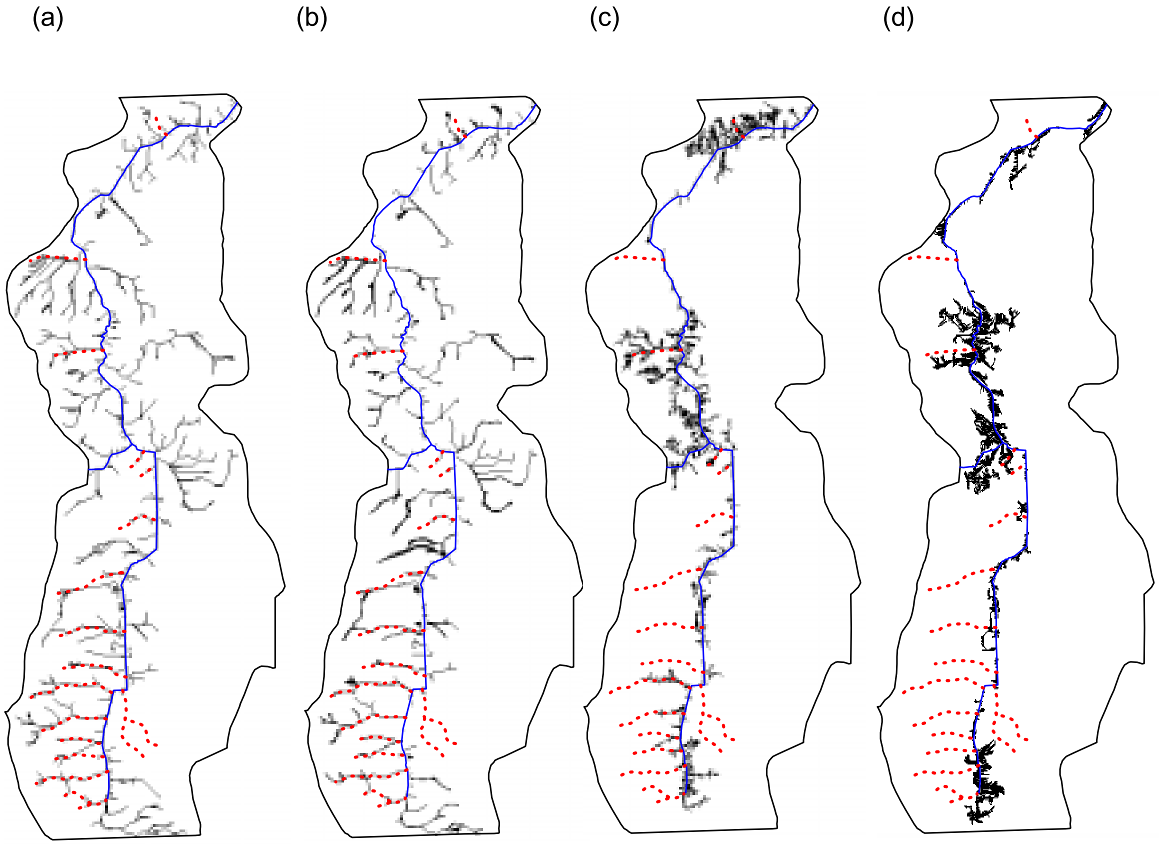


Figure 8.

Figure 9.

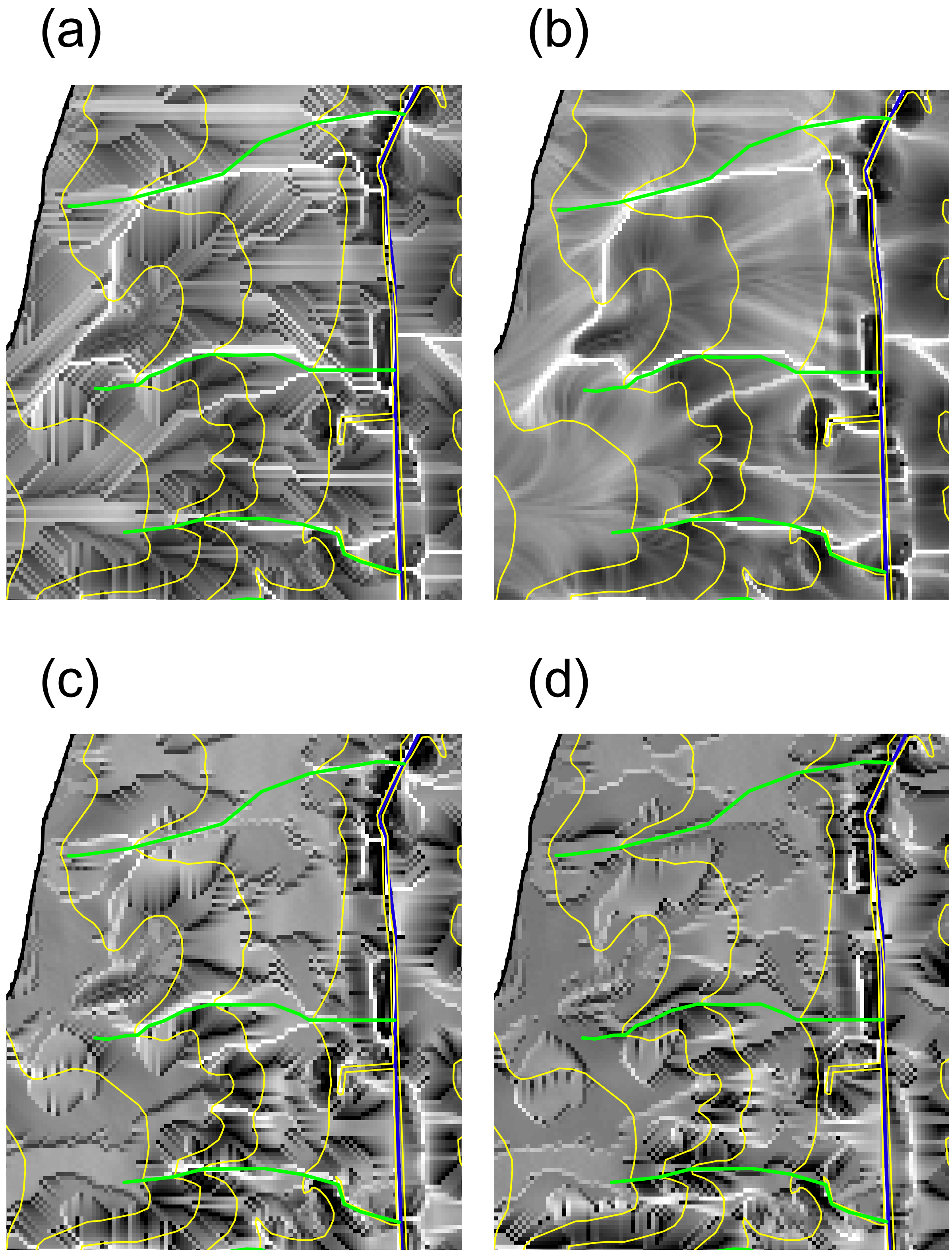
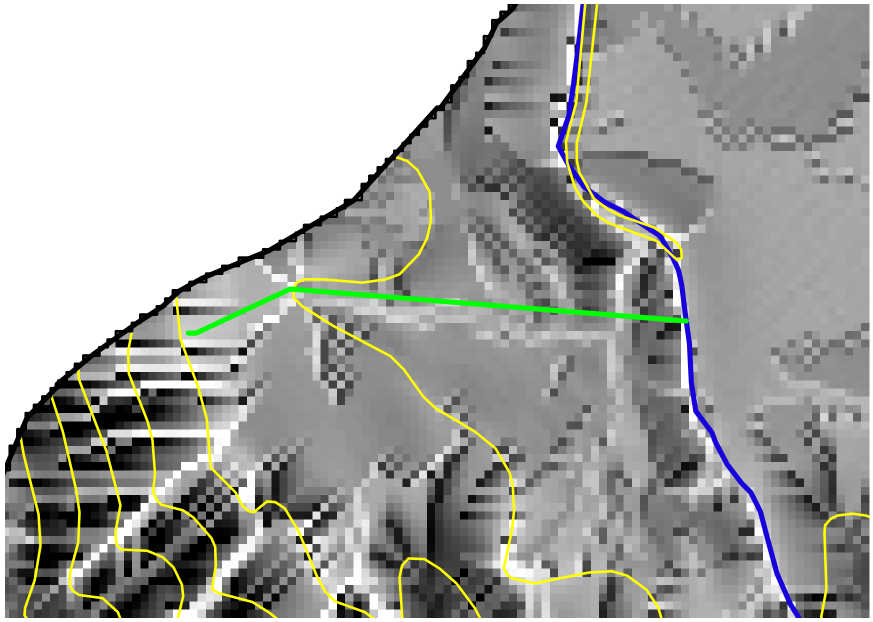


Figure 10.

a)



b)

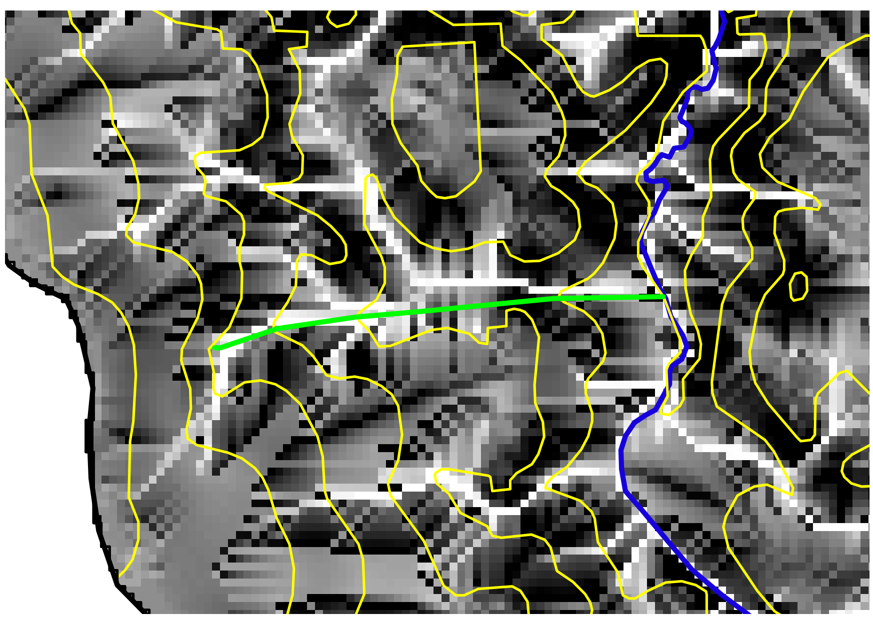


Figure 11

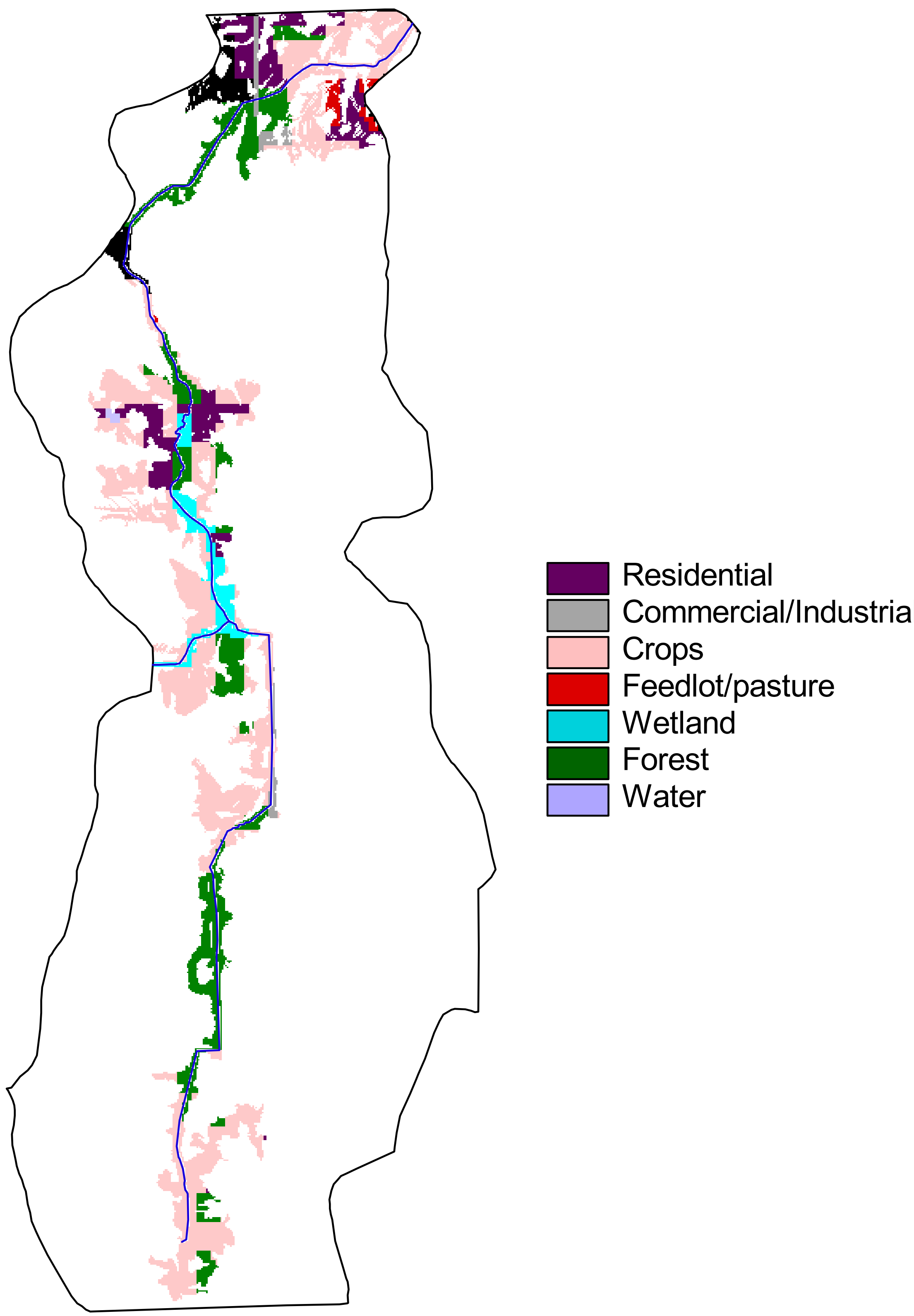

Residential
Commercial/Industrial
Crops
Feedlot/pasture
Wetland
Forest
Water

Figure 12.