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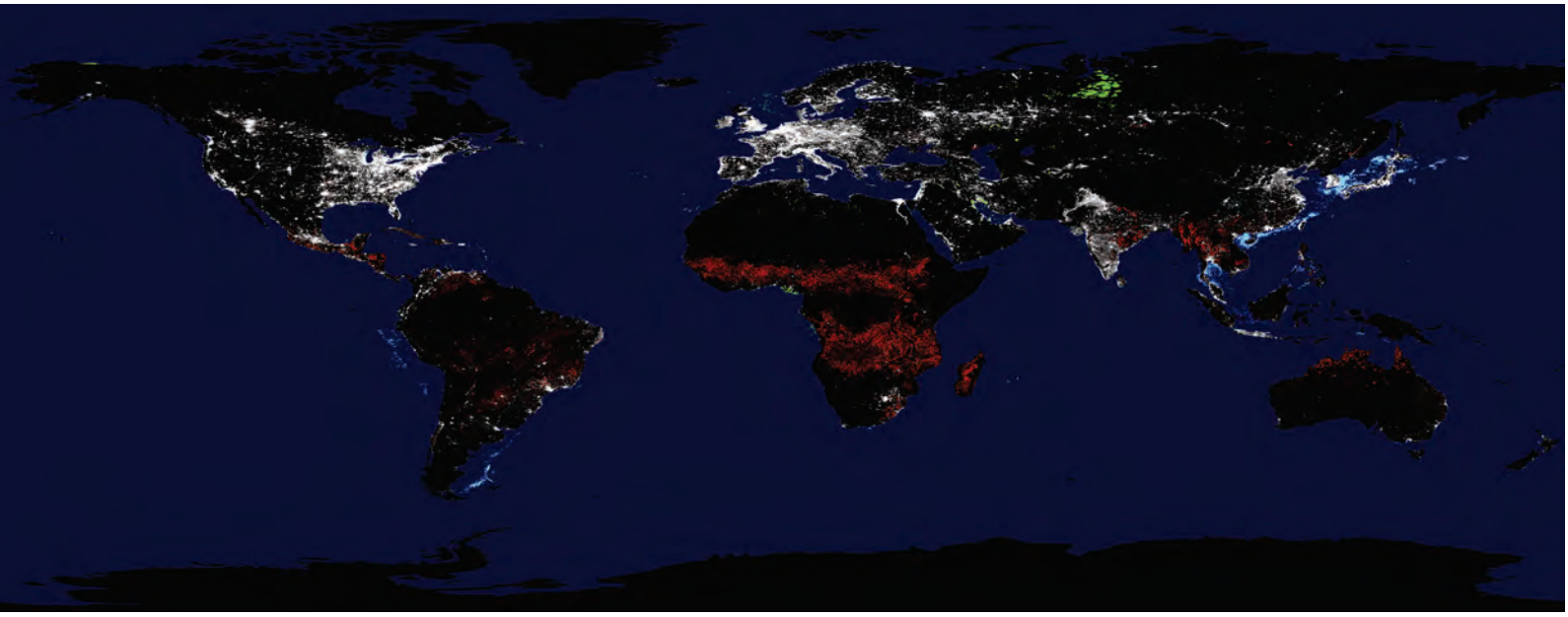
Landscape and Predictive Tools

A Guide to Spatial Analysis for Environmental Assessment



United States Environmental Protection Agency
Office of the Science Advisor, Risk Assessment Forum

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ABSTRACT

This landscape and predictive tools methods manual, developed collaboratively by U.S. EPA's Office of Water, Office of Research and Development, Regional Offices and others, describes the purpose, rationale, and basic steps for using landscape and predictive tools for Clean Water Act monitoring, assessment and management purposes. Landscape and predictive tools are needed both to guide efficient filling of monitoring gaps and to prioritize our protection and *rehabilitation* actions. This should yield better protection for high quality waters and quicker, more cost-effective *restoration* of *impaired* waters.

We have organized this method guidance document into four sections: (I) Introduction to Landscape and Predictive Tools; (II) Geographic Frameworks, Spatial Data, and Analysis Tools; (III) Examples and Case Studies; and (IV) Gaps and Needs for Research and Applications. In addition, the extensive Toolbox provides links to and short descriptions of a wide range of easily accessed data sets and analytical tools.

This guidance stresses simultaneous use of *matched* (or paired) landscape and *in situ* data for empirical modeling to enhance our predictive capabilities and encourage science-based *targeting* and *priority* setting. Landscape and predictive tools have a wide range of current and potential applications including criteria and standards development, problem identification and prevention, prioritization and targeting of rehabilitation, and advancing science, education, and society's ability to effectively manage aquatic and terrestrial resources.

Particularly valuable assets are models that combine *in situ* field measurements and landcover—thus, providing us with landscape and predictive tools for many water quality programs. For example, empirical models that provide a predictive capacity are

including mapped areas such as ecoregions or other appropriate classification approaches, to establish realistic areas for analysis and extrapolation, (2) use "wall-to-wall" landscape and other data to document stress gradients, (3) construct empirical relationships or models linking landscape and other stress indicators to *in situ* response, and (4) use these relationships to extrapolate to places lacking *in situ* data. Estimating the condition of places lacking data can greatly expand the usefulness of our limited site-specific data. Regular use of landscape and predictive tools can support comprehensive, systematic priority setting and targeting for monitoring, rehabilitation and prevention actions. Using these tools as a matter of course will require ongoing commitment to training, continued collaborative development of applications, new techniques and data, and consistent effort to bring these scientific advances to bear on our water quality monitoring, assessment, rehabilitation, and protection efforts.

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Nighttime Lights of the World

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In this section, we describe the geographic frameworks most commonly used to research, monitor, assess, and manage aquatic resources. The success of these activities depends on a sound understanding of the underpinnings, strengths, and limitations of the geographic frameworks used. We also discuss how frameworks can be combined to maximize the ability to extrapolate. We emphasize simultaneous use of ecological regions and watersheds as an example of an appropriate application of using differing geographic frameworks as landscape and predictive tools.

What is in this chapter? A review of geographic frameworks commonly used in landscape analysis with examples and discussions of their strengths and weaknesses. A geographic framework is the organizing structure for a database or analytic process having spatially linked attributes such as latitude and longitude, relative position as along a stream, or designated class such as a county, watershed, or waterbody size. Different geographic frameworks are designed for different purposes at different spatial extents and levels of resolution, and therefore combining them and interpreting results requires logical rigor.

The most commonly used geographic frameworks (or spatial classifications) in aquatic resource science are ecoregions, watersheds/basins, and hydrologic units (HU) (often referred to as *HUCs* because of the assigned hydrologic unit code assigned by the U.S. Geological Survey [USGS]). All three types of tools provide maps or geographic spatial models of areas or parts of the Earth's surface sharing common characteristics expected to be useful for understanding patterns in the structure, function, and responses of ecosystems—especially in cases where quantitative, site-specific data are insufficient.

The three common geographic frameworks discussed below are widely and frequently used to do the following:

- Predict patterns or thresholds in natural ecosystem gradients and in anthropogenic stressor/disturbance gradients.
- Plan and target monitoring to maximize efficient use of limited fiscal resources.
- Develop conservation plans to protect highly valued natural resources.
- Identify and select ecologically appropriate reference sites.
- Interpret patterns in ecological data rationally.
- Extrapolate monitoring results to areas lacking *in situ* data.
- Develop regional adaptations of national water quality standards that are based on ecological units versus political units.
- Focus rehabilitation efforts in a cost-efficient and ecologically responsive manner.

5.1. ECOREGIONS

5.1.1. Description

Ecoregions (or ecosystem regions) are defined as areas of relative homogeneity in ecosystems or relationships among organisms and their environments. Ecoregion maps depict areas within which the aggregates of all terrestrial and aquatic ecosystem components are different from or less variant than those in other areas (Omernik and Bailey, 1997). In explaining the process of defining ecoregions, Wiken (1986) stated:

> Ecological land classification is a process of delineating and classifying ecologically distinctive areas of the earth's surface. Each area can be viewed as a discrete system which has resulted from the mesh and interplay of the geologic, landform, soil, vegetative, climatic, wildlife, water, and human factors which may be present. The dominance of any or a number of these factors varies with the given ecological land unit. This approach to land classification can be applied incrementally on a scale related basis from very site specific ecosystems to very broad ecosystems.

Although few ecoregions specifically follow Wiken's vision, the following geographic frameworks have been called or used as ecoregions:

- Ecoregions (Bailey, 1976, 1998).
- Ecoregions (Omernik, 1987; U.S. EPA, 2011).
- Major Land Resource Areas (USDA, 2006).
- ECOMAP terrestrial ecological units (Keys et al., 1995; Cleland et al., 2007).
- Aquatic Ecological Units (Maxwell et al., 1995).
- Ecological Regions (Commission for Environmental Cooperation [CEC], 1997).
- Ecoregions (Hargrove and Hoffman, 1999, 2004).
- Terrestrial Ecoregions (Ricketts et al., 1999).
- Freshwater Ecoregions (Abell et al., 2000, 2008).
- Common Ecological Regions (McMahon et al., 2001).
- Terrestrial Ecosystems (Sayre et al., 2009).

For in-depth discussions and comparisons of these and other ecoregion-type frameworks, see McMahon et al. (2001), Gallant et al. (2004), Loveland and Merchant (2004), and McMahon et al. (2004). The last three of these papers and other articles on ecoregions appeared in a special issue of *Environmental Management* titled *Ecoregions for Environmental Management*.

5.1.2. Strengths and Limitations

Most of the above frameworks were compiled by different individuals or groups, using different methods, and for different purposes, which has led to misuse and misunderstanding of ecoregions. Frustrated by a lack of conformity among resource management agencies regarding a national framework of ecological regions, the U.S. Government Accountability Office (GAO, 1994) stated that a common framework is necessary for management of ecosystems versus specific resources. Recognizing this problem identified by the GAO report, an interagency group called the National Interagency Technical Team (NITT) was formed, and, together with an interagency steering committee, created a Memorandum of Understanding focusing on developing a

common framework of ecological regions of the United States (McMahon et al., 2001). Many involved in this interagency effort were reluctant to delve into the reasons for the differences in the existing frameworks and to evaluate them against the broader purpose of a common framework. However, the NITT did agree on a goal that fit the objectives of the GAO report—to identify regions of similarity in the mosaic of biotic, abiotic, terrestrial, and aquatic ecosystems with humans considered a part of the biota (McMahon et al., 2001; Omernik, 2004). The purpose of this common framework is identical to that of the CEC ecological regions and the U.S. Environmental Protection Agency (EPA) ecoregions—to provide a geographical tool to aid environmental resource managers with different responsibilities for the same geographic areas to integrate their research, monitoring, assessment, and management activities.

Because of the breadth of this purpose (addressing ecosystems in the broadest sense of the word), frameworks that were developed for specific objectives are likely to be more effective for management concerns regarding those specific purposes and less effective for integrating activities of all ecosystems (aquatic, terrestrial, biotic, and abiotic). For example, the World Wildlife Fund terrestrial ecoregions, which used EPA ecoregions that were split or merged to distinguish particular assemblages of species or unique habitats, might be more appropriate for assessing terrestrial biodiversity issues. Another framework, the Natural Resources Conservation Service (NRCS) Major Land Resource Regions (USDA, 2006)—which is based primarily on soil characteristics from state general soil map units, and secondarily on land use, climate, physiography, vegetation, water resources, and geology—was intended to address soil capacities and agricultural potential and is ideally suited for those subjects. However, many in the NRCS recognize the value of the NITT, EPA, and CEC frameworks and have collaborated with EPA and other agencies in developing Level III and IV ecoregion maps for states (e.g., McGrath et al., 2002; Thorson et al., 2003; Daigle et al., 2006).

Some organizations, including the U.S. Department of Agriculture (USDA) Forest Service (Keys et al., 1995; Maxwell et al., 1995), and the World Wildlife Fund (Ricketts et al., 1999; Abell et al., 2000, 2008), have developed separate frameworks for terrestrial and aquatic ecosystems. Although these mapped frameworks are useful for specific aspects of ecosystems in some areas, none is ideally suited for the monitoring,

assessment, research, inventory, and management of entire ecosystems. The frameworks of Maxwell et al. (1995) and Abell et al. (2000, 2008), which were designed to address aquatic ecosystems, are problematic because they are based on hydrologic units. As such, these maps might be useful in identifying potential patterns in distributions of some fish species (Jelks et al., 2008), but they are of little help in addressing true fish species pools (e.g., McGarvey and Hughes, 2008; McGarvey and Ward, 2008), or patterns in physical and chemical habitat or benthic macroinvertebrates. Aquatic insects, for example, spend their lives in both the aquatic and terrestrial environments, and unlike fish, their distributions and abundances are not restricted by drainage basins.

Misunderstanding the general purpose or ecosystem nature of ecoregions has led to the development of frameworks that are called ecoregions but were in fact designed for addressing a specific ecosystem component. True ecoregions are not intended to replace mapped frameworks of vegetation, geology, soil, water quality, fish distributions, climate, landcover, or physiography, nor should they replace frameworks that consider some but not all these characteristics. Hence, patterns in characteristics such as nutrient concentrations in streams are better explained by spatial differences in land use (Omernik, 1977) than by ecological regions (Wickham et al., 2005). Likewise, patterns in fish and benthos distributions might be better explained by patterns in current and historical river basins (Hocutt and Wiley, 1986; Hughes et al., 1987) or by habitat gradients (Hawkins et al., 2000b; Hughes et al., 2006). However, when assessing and managing aquatic ecosystems, one must consider *all* aspects of aquatic ecosystems, including the biota and chemical and physical habitat (Karr, 1993, 1995; Yoder, 1995). EPA ecoregions have proven useful for many of these purposes (e.g., Larsen et al., 1988; Hughes et al., 1994; Davis et al., 1996). For example, Tennessee has used Level III and IV EPA

Reasons for Disagreement About How to Delineate Ecoregions

- Different definitions of *ecosystems.*
- Failure to embrace ecosystem holism.
- Ecoregion and boundary complexity.
- Bias toward a single characteristic.
- Rule-based vs. weight-of-evidence approach.
- Whether watersheds and hydrologic units are ecoregions.
- Investment in existing frameworks and resistance to change.

ecoregions to assess habitat quality of least disturbed streams (Arnwine and Denton, 2001b), develop regional nutrient criteria (Denton et al., 2001), and develop regional biological criteria (Arnwine and Denton, 2001a).

Because of the nature of their development, ecoregions depict general patterns in combinations of ecosystem components rather than the degree to which a single component matches each region. For example, Larsen et al. (1988) found that the patterns of single chemical parameters in Ohio streams were seldom associated with EPA ecoregions. However, a principle components analysis of combinations of all the chemical characteristics sampled, with a combination of components comprising nutrient richness on one axis and a combination of components comprising ionic strength on the other, revealed a strong ecoregion pattern (Larsen et al., 1988; see Figure 5-1).

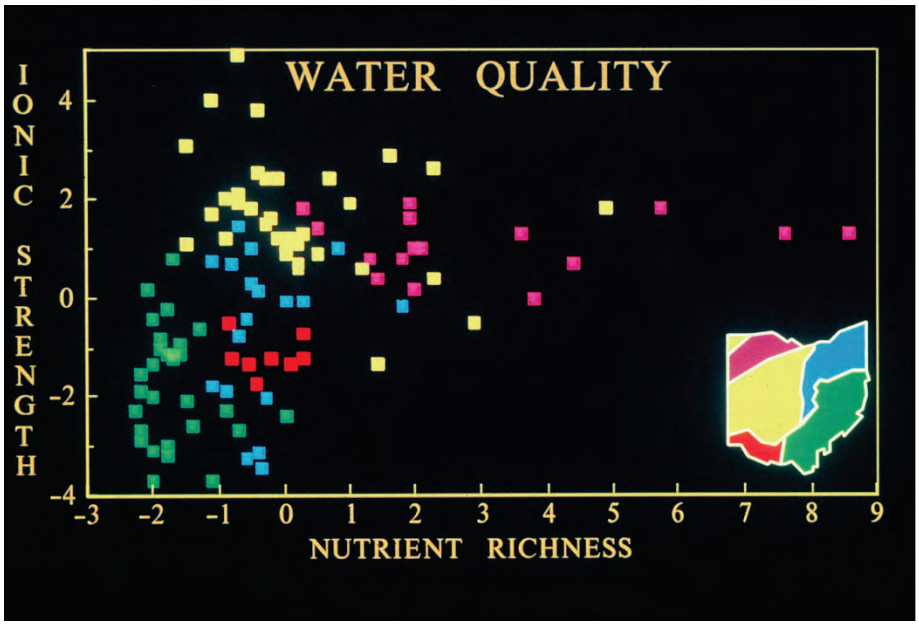

FIGURE 5-1

Ohio Ecoregional Patterns in Nutrient Richness and Ionic Strength Variables in Least-Disturbed Watersheds as Indicated by Principal Components Axis Scores for Each. Each square color corresponds to a site in an ecoregion of the same color on the index map.

Source: Larsen et al. (1988), Griffith et al. (1999).

Ecoregions are effective for integrating ecosystem management activities among resource management agencies and programs because all agencies and programs are using the same *pieces of the puzzle*, but might merely be aggregating them differently. The Tennessee Department of Environment and Conservation, for example, did not use the same groups of Level III and IV EPA ecoregions for each of their applications; they used only those useful for explaining patterns of their water quality data (Arnwine and Denton, 2001b). Other states such as Iowa have used both Level III and IV EPA ecoregions to locate long-term monitoring sites to monitor water quality trends as pollution control practices are implemented, farming practices evolve, and watersheds undergo urban and industrial development (IDNR, 2001). However, Iowa fish distributions and assemblage conditions showed little relationship to EPA ecoregions (Heitke et al., 2006). On the other hand, Arkansas fish assemblages showed distinct ecoregional patterns (Rohm et al., 1987). Arkansas also found Level III EPA ecoregions useful for developing and evaluating water quality standards (Omernik and Griffith, 1991) and has subsequently adopted the framework for its multiagency Comprehensive Wildlife Conservation Strategy (http://www.wildlifearkansas.com/strategy.html). Other organizations have found EPA ecoregions useful for applications that require consideration of aquatic and terrestrial ecosystems. The North American Bird Conservation Initiative uses EPA ecoregions as a framework for biological conservation research and planning (U.S. NABCI Committee, 2000). The Nebraska Game and Parks Commission and Georgia Department of Natural Resources used Level III and IV EPA ecoregions to classify areas of bird occurrence (Farrar, 2004; Schneider et al., 2010).

Ecoregions are increasingly being used for national-level research and assessment projects of environmental resources. The U.S. Geological Survey uses Level III EPA ecoregions to stratify its land cover project that evaluates the rates, trends, causes, and consequences of land use and land cover change in the conterminous United States (Drummond and Loveland, 2010; Napton et al., 2010; Sleeter et al., 2011). The EPA has adapted Level II ecoregions to report the results of the Wadeable Streams Assessment (U.S. EPA, 2006; Paulsen et al., 2008) and the

National Lake Assessment (U.S. EPA, 2009). Other examples of ecoregion applications can be found online at http://www.epa.gov/wed/pages/ecoregions/links.htm.

5.2. SINGLE-PURPOSE FRAMEWORKS

5.2.1. Description

Unlike ecoregions, which were designed to identify areas in which there is similarity in the mosaic of all ecosystem components (aquatic, terrestrial, biotic, and abiotic), a number of single-purpose frameworks have been developed to delineate areas in which there is similarity in a specific geographic phenomenon or subject of interest to environmental resource management. Examples of such maps include those that have been developed for acid sensitivity of surface waters (Omernik and Powers, 1983; Omernik and Griffith, 1986; Omernik et al., 1988a) stream nutrient levels (Omernik, 1977), lake trophic state (Omernik et al., 1988b; Rohm et al., 1995), land use/landcover (Loveland et al., 1991; Vogelmann et al., 2001), and agricultural practices and products (USDA, 1999).

As with ecoregions, pattern analysis is necessary in developing many of these single-purpose maps. For example, in mapping alkalinity of surface waters, one must examine the patterns of alkalinity values together with maps of spatial characteristics (including soil, geology, vegetation, and land use) that can be associated with regional differences in alkalinity. At the same time, one must ensure that the watershed sizes used to help detect patterns are consistent with the spatial heterogeneity and homogeneity of the spatial characteristics affecting regional differences. For instance, by evaluating patterns of surface water alkalinity values in very small watersheds in the mountainous western United States, abrupt changes in values can be detected that follow differences in geology. On the other hand, because of soil buffering capacity in most cropland regions, alkalinity values tend to be consistently high across broad areas. In such areas, the land use/alkalinity value association can be detected from patterns in water quality data associated with watersheds of varying sizes.

Some single-purpose maps, such as those showing agricultural census data, use dot patterns and county frameworks to show patterns in agricultural characteristics (e.g., USDA, 1999). In the central and eastern United States, where most counties are small

and of relatively similar size, general patterns in characteristics shown by county can be seen. However, illustrating these patterns through use of dots is often more effective if the dots are in the agricultural parts of each county. This is especially true in much of the western United States where there are very large counties, many of which include small areas of intensive agriculture and much larger areas of forest, rangeland, or desert. A similar dichotomy occurs with maps of fish species occurrences between dot maps (e.g., Lee et al., 1980) versus HU or basin maps (e.g., Hocutt and Wiley, 1986). As with agriculture, the dot maps more accurately illustrate the presence (or probable absence) of a species than do basin or HU maps that imply that a species is present throughout the larger map unit, rather than just the headwaters or tidal reaches.

5.2.2. Strengths and Limitations

The obvious value of single-purpose maps is that they illustrate spatial patterns of particular aspects of interest to resource managers and scientists, thereby allowing them to structure their management and research according to regional patterns and trends in those aspects. The limitations of these frameworks are the same as with any map, whether of land use, alkalinity of surface waters, rock type, or vegetation—they are only spatial representations of particular characteristics at a scale smaller than 1:1. Moreover, many if not most of the characteristics mapped vary temporally. Hence, each of these maps should be considered as representations during a particular time.

5.3. WATERSHEDS

5.3.1. Description

Watersheds (also basins and catchments) are topographic areas in which surface and ground water drain to a specific point (Omernik and Bailey, 1997; Griffith et al., 1999). Webster's Dictionary (Merriam-Webster, 1986) defines a watershed as "a region or area bounded peripherally by a water parting and draining ultimately to a particular watercourse or body of water." Both definitions above are essentially the same and are unambiguous.

5.3.2. Strengths and Limitations

As defined above, watersheds have been, and will continue to be, powerful tools for water resource managers and scientists for associating natural and anthropogenic characteristics with water quality and quantity. Where watersheds can be defined and are in mesic and hydric areas, their downstream points reflect the aggregate of the characteristics upgradient from each point.

Nonetheless, in many parts of the country, watersheds are either difficult or impossible to define (Hughes and Omernik, 1981). Regions of karst topography, continental glaciation, extremely flat plains, deep sand, and xeric climates fall into this category (Omernik and Bailey, 1997; Currens and Ray, 2001). In many xeric regions of the country where apparent watersheds can be defined and *influent* streams predominate (where streams feed the ground water rather than where the ground water feeds the streams), topographic watersheds do not always encompass the same integrating processes as in mesic and hydric areas (Strahler, 1975; Omernik and Bailey, 1997). Such streams not only lose water, but they also become markedly more saline and enriched than effluent streams as a result of evapotranspiration, and they often flow through surficial mineral deposits that are remnants of pluvial periods. Also, the usefulness of true topographic watersheds is lessened in places where water has been diverted from one drainage basin to another and where flows are dominated by wastewater effluents or irrigation return flows, which is common in xeric areas of the western United States. Although one could argue that the watershed boundaries could be modified to account for inputs from diversions, this can be difficult to impossible because water diversions are rarely constant, can be reversed by pumping, and are inconsistent with precipitation.

5.4. HYDROLOGIC UNITS (HUS) AND HYDROLOGIC UNIT CODES (HUCS)

5.4.1. Description

Hydrologic units were developed initially from the USGS digital framework (Seaber et al., 1987), and the EPA River Reach File. The framework is hierarchical and shows “drainage hydrography, culture, and political and hydrologic unit codes (HUCs)” (Seaber et al., 1987). The system divides the United States into 21 major regions,

which are subdivided into 222 subregions and 352 accounting units, and finally subdivided into 2,149 cataloging units. These hydrologic units are commonly known as *HUCs,* although the hydrologic unit codes are merely identifiers for the units at each hierarchical level. In this section, all hydrologic units are called *HUs*. The 21 major regions contain either the drainage area of a major river (only 2 cases: Ohio, Upper Mississippi) or the combined drainage areas of a series of rivers (19 cases, 3 of which are based on political units: Alaska, Hawaii, Puerto Rico). Each "subregion includes the areas drained by a river system, a reach of a river and its tributaries in that reach, a closed basin(s), or a group of streams forming a coastal drainage area" (Seaber et al., 1987). The accounting units nest within or are the same as subregions, and each of the cataloging units (8-digit HUs) is an area representing part or all of a drainage basin, a combination of drainage basins, or a distinct hydrologic feature (Seaber et al., 1987).

Subsequent to the development of the framework by Seaber et al. (1987), the USDA-NRCS, in collaboration with other federal agencies and many state resource management agencies, began an effort to delineate smaller, more detailed, 10- and 12-digit HUs (Federal Geographic Data Committee, 2004; Berelson et al., 2004; Eidson et al., 2005). Whereas 2,146 8-digit HUs cover the United States, it is estimated that roughly 22,000 10-digit and 160,000 12-digit units will be delineated for the country (Federal Geographic Data Committee, 2004).

5.4.2. Strengths and Limitations

The major strength of HUs that are also watersheds is, of course, the same as that previously stated for watersheds. The value of HUs is that such a framework provides a wall-to-wall national set of terrestrial polygons of roughly comparable size at each subdivision. As such, the framework is often used for depicting spatial pattern of resources (e.g., NatureServe, 2005; Jelks et al., 2008), if not connectivity. However, equally useful and meaningful polygons include hexagons (Rathert et al., 1999) and quadrangles (McAllister et al., 1986). It is essential to understand that usually fewer than half of HUs comprise true watersheds regardless of the level of subdivision.

A major limitation of HUs lies in the mistaken statement of their intended use, which according to Seaber et al. (1987) is to provide "a standard geographic and

hydrologic framework for water-resource and related land-resource planning." The logic of using these units for these purposes is questionable because they are largely unrelated to patterns of geographic characteristics that are associated with spatial differences in water quality and quantity (Omernik and Bailey, 1997; Griffith et al., 1999; Omernik, 2003; Brenden et al., 2006; Hollenhorst et al., 2007). For example, HUs tend to include dissimilar ecoregions, as do watersheds.

A second major limitation of HUs is the common misconception that they are watersheds. Although Seaber et al. (1987) did not define HUs as true watersheds, many believe that watersheds and HUs are synonymous (e.g., USFWS, 1995; Jones et al., 1997; Ruhl, 1999; Alexander et al., 2000; Graf, 2001). However, most HUs are **not** true watersheds. Because streams are *linear* features rather than spatial features, it is impossible to map the watersheds of an area such as a continent, country, or state so that it is completely covered with watersheds of similar size (Omernik, 2003). Omernik (2003) reported that national maps of HUs (6-digit, 8-digit, or any other hierarchical level) contain only about 45% true watersheds (10% in the case of major regions). This inherent problem with the HU framework at the 8-digit level has also been demonstrated for Texas (see Figure 5-2, Omernik, 2003). Most HUs are watershed segments, or watershed groups that do not serve the critical purpose of true watersheds. For these reasons, the terms *HUs* and *watersheds* should not be used interchangeably. Attempts have been made to code HUs to allow one to distinguish true watersheds (Verdin and Verdin, 1999). However, none allows the determination of true watersheds at all levels (e.g., 6-, 8-, 12-, or *n*-digit) directly, including those of Verdin and Verdin (1999), Seaber et al. (1987), the Federal Geographic Data Committee (2004), and Eidson et al. (2005). Instead, one must first employ data from digital elevation maps, the national elevation data set, or models (Brenden et al., 2006; Hollenhorst et al., 2007). Unfortunately, even some developers of the 12-digit HUs (e.g., Berelson et al., 2004), who recognized the inaccurate perception and relationship that is perpetuated by labeling HUs as watersheds, have not attempted to rectify this problem by appropriate labeling, thereby furthering the inaccurate perception.

Because of the importance of rectifying this common misperception, we illustrate the limitations of HUs by also examining the 8-digit HUs within the Columbia basin of

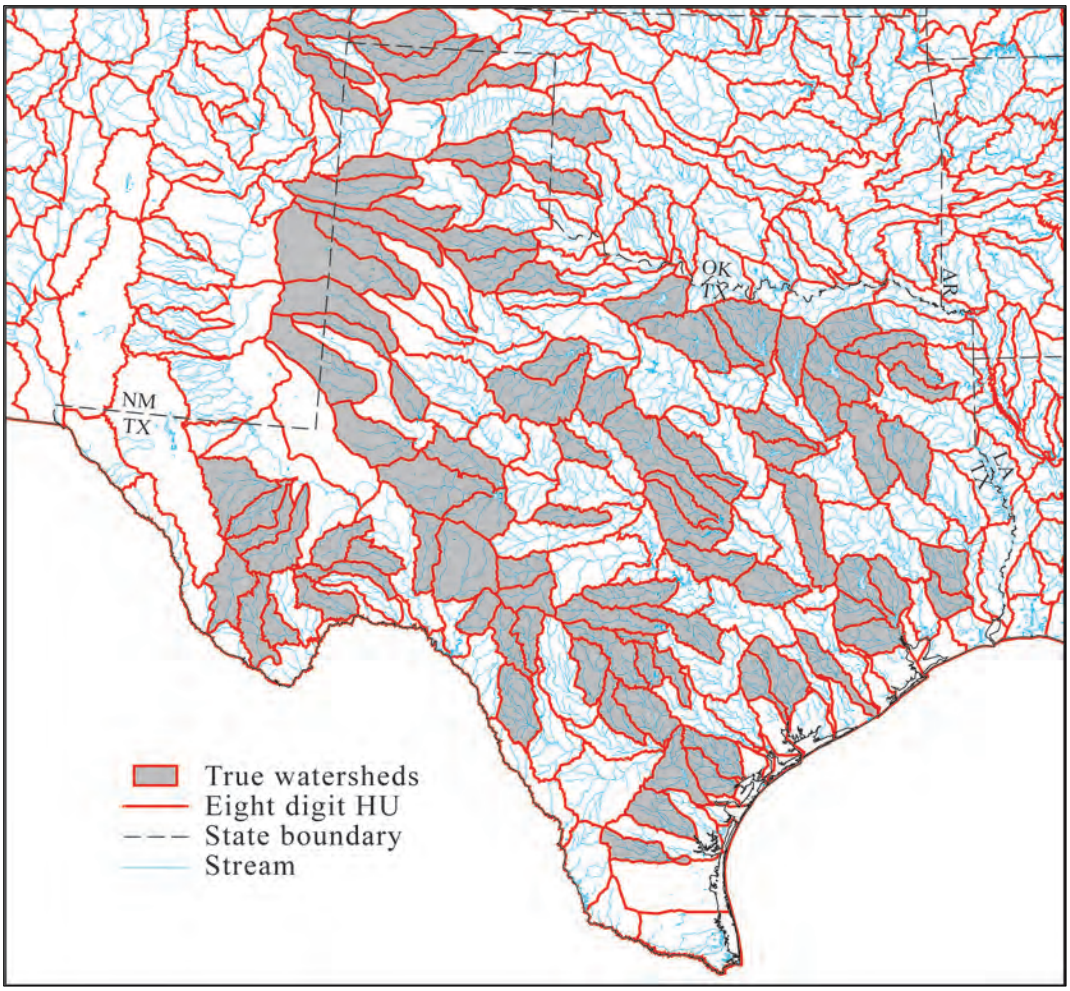

FIGURE 5-2

Eight-Digit HUs of Texas that are True Watersheds. Note, only 48% of the HUs are watersheds.

Source: Omernik (2003).

the Pacific Northwest (see Figure 5-3). Only 58% of these HUs are true watersheds (see Figure 5-4). If all HUs were true watersheds one would expect the water quality and flow regime at downstream points of HUs within the same ecoregion to be similar as compared to those of adjacent ecoregions where conditions are markedly different. For example, four 8-digit HUs lie completely or nearly completely within the Columbia Plateau ecoregion which is steppe, and flanked by forested, mountainous ecoregions (see Figure 5-5; Thorson et al., 2003). However, only two (B and C) of the four 8-digit HUs are true watersheds (see Figure 5-6). One of the HUs (A) is a downstream segment of the Columbia River and drains large parts of northeastern Washington, northern Idaho, northwestern Montana, and southeastern British Columbia. The other HU (D) is a downstream segment of the Snake River that drains eastern Oregon, most of Idaho, and parts of Nevada and western Wyoming. Like watersheds, the quality and quantity of water at the downstream segments of HUs A and D (which drain large forested ecoregions) will be vastly different from those of HUs B and C (which drain largely sagebrush steppe). There are 8-digit HUs that are true watersheds completely or nearly completely within specific Level III ecoregions of the Columbia Plateau (see Figure 5-7). Data from these types of HUs are useful for determining patterns of water quality and flow regime in the Columbia Plateau.

The problem with HUs can also be seen at the 12-digit level, using South Carolina as an example. Of the 986 12-digit HUs completely or partially in the state, only 47% are true watersheds (see Figure 5-8). If all 12-digit HUs were true watersheds, one would expect the water quality and flow regime at the downstream point of all HUs within an ecoregion to be relatively similar and somewhat different than those of HUs within other ecoregions where factors affecting flow and water quality differ (see Figure 5-9). Several 12-digit HUs are within the Carolina Slate Belt Level IV ecoregion or the Sand Hills Level IV ecoregion (see Figure 5-10). Streams in the Carolina Slate Belt with watershed areas <20 km^2 tend to dry up annually because the region contains some of the lowest water-yielding rock in the Carolinas (Griffith et al., 2002). In the Sand Hills, on the other hand, such streams rarely dry up or flood because of the high storage capacity of the sand aquifer (Griffith et al., 2002). Hence, the 12-digit HUs that are true watersheds completely within one of these ecoregions will

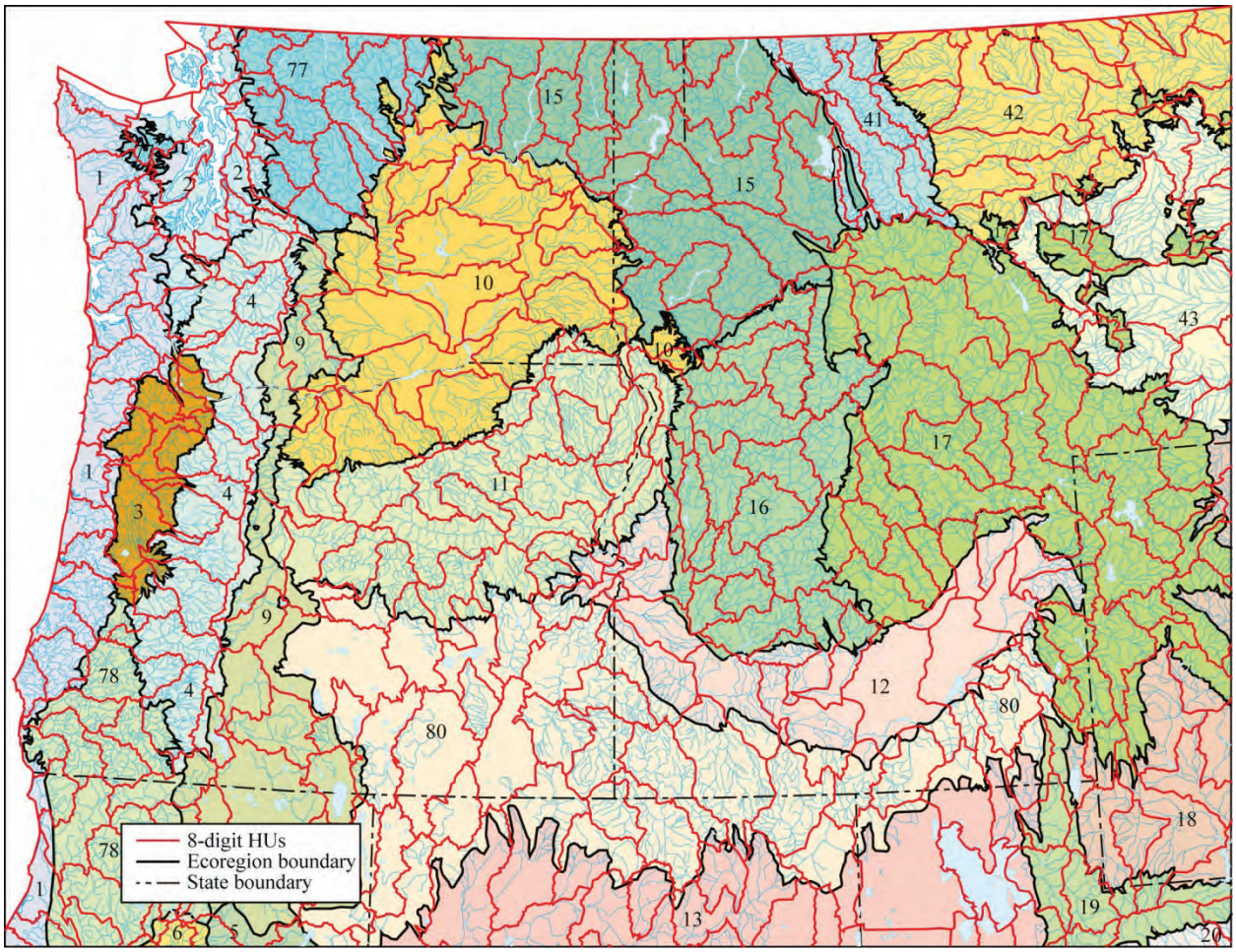

FIGURE 5-3

Level III Ecoregions and 8-Digit HUs of the Pacific Northwest

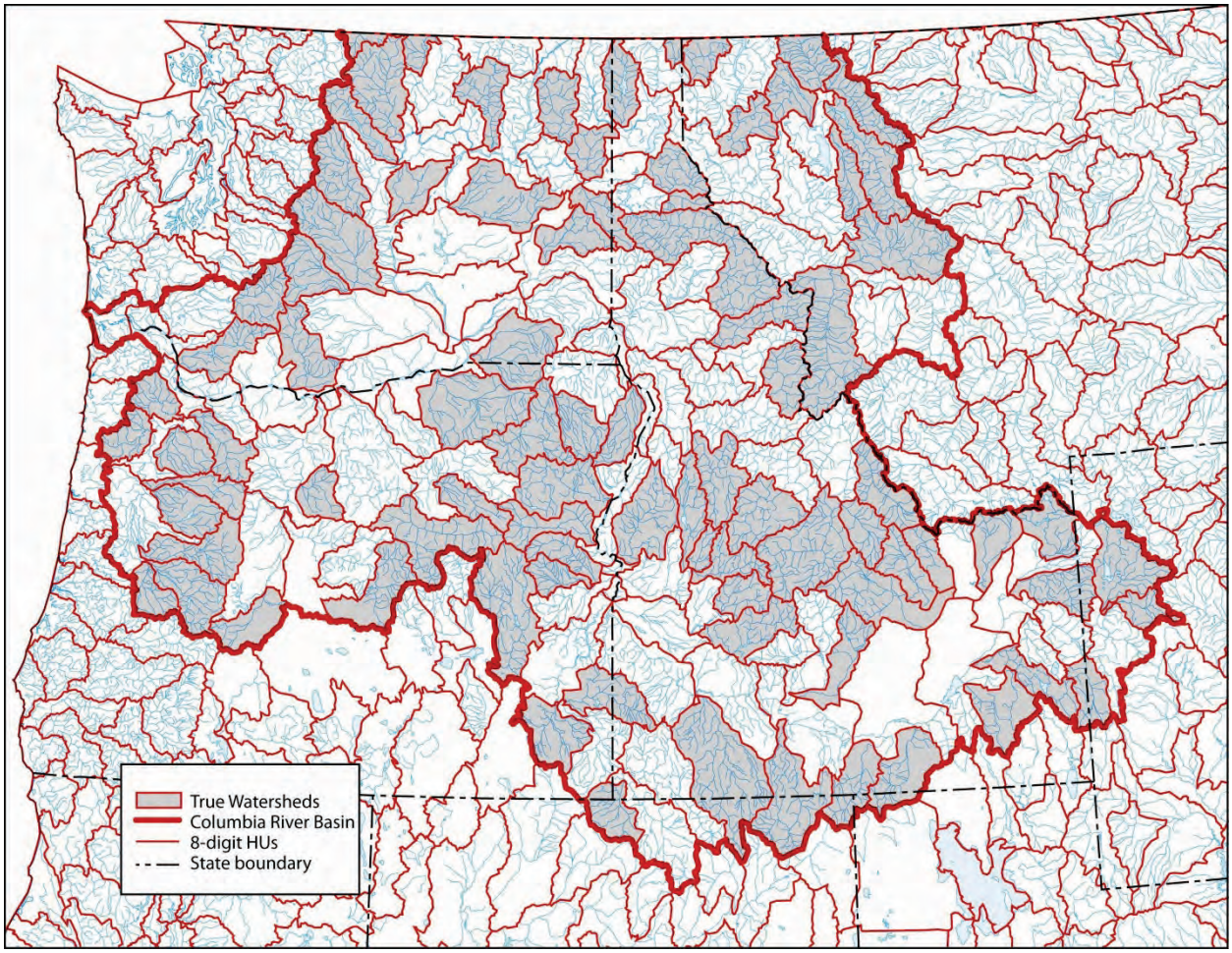

FIGURE 5-4

Eight-Digit HUs that are True Watersheds within the Columbia Basin.
Note that only 58% of the HUs are True Watersheds.

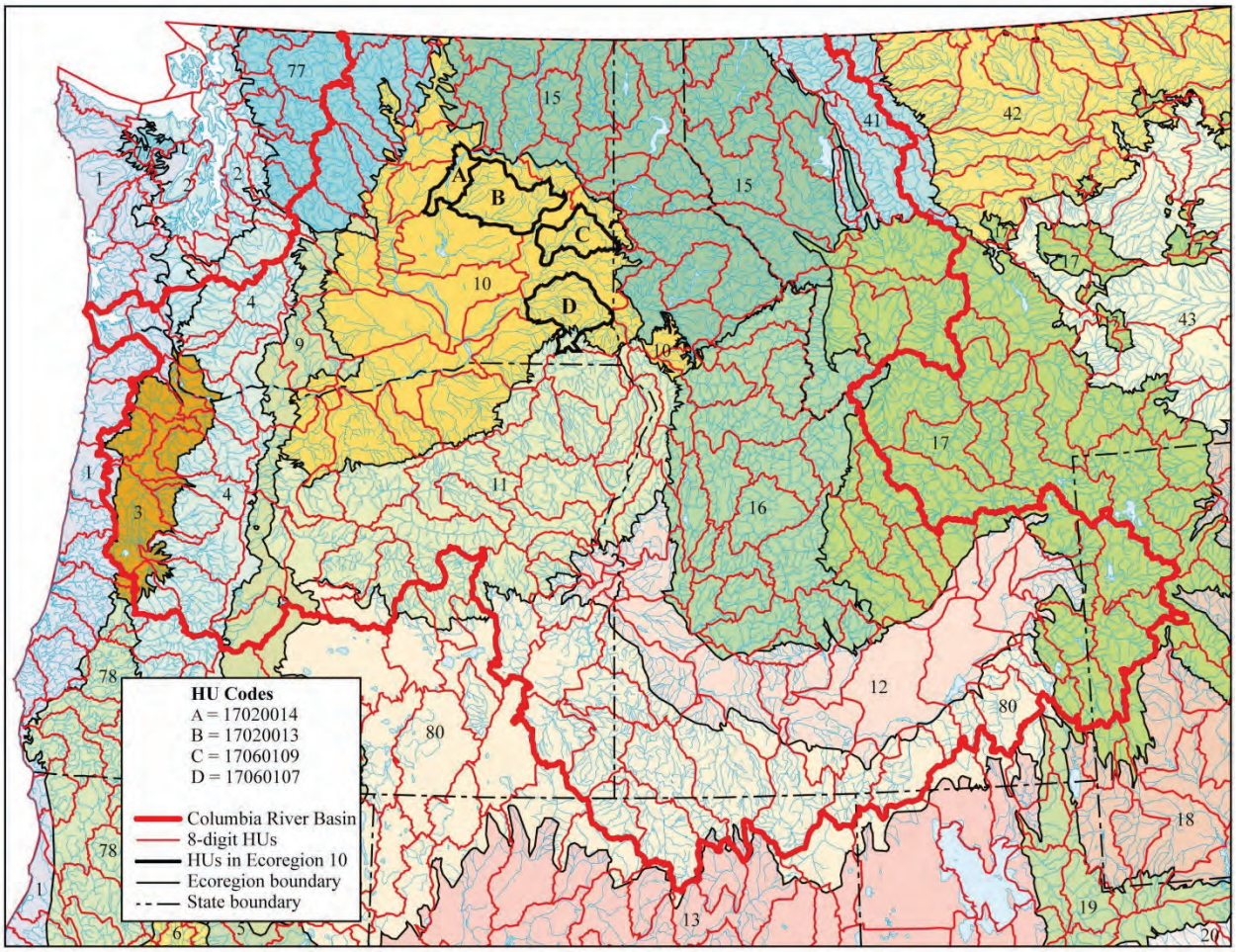

FIGURE 5-5

Four 8-Digit HUs (A, B, C, and D) in the Columbia Plateau Level III Ecoregion (10)

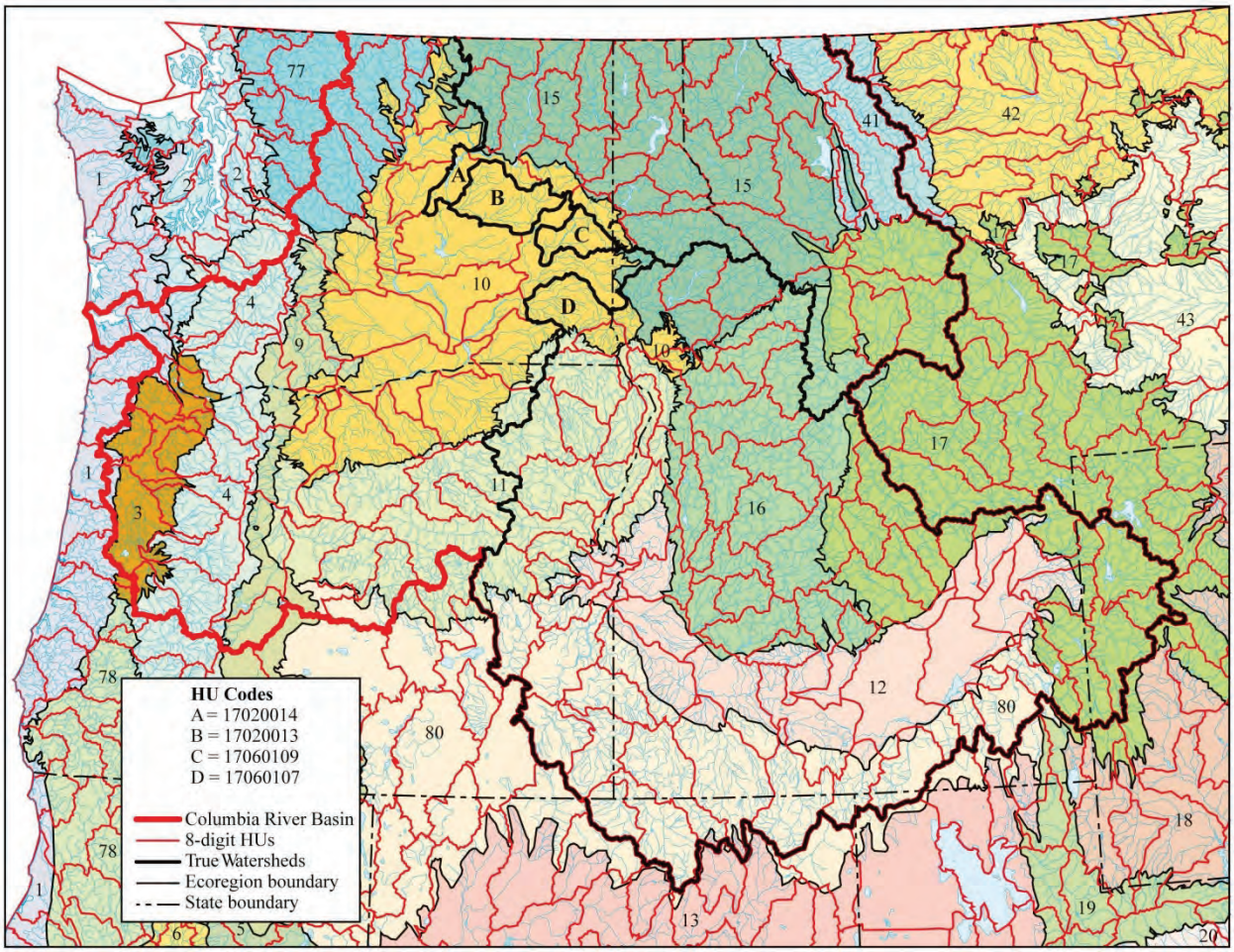

FIGURE 5-6

True Watersheds Associated with Downstream Points in HUs A, B, C, and D. Note that B and C are true watersheds, whereas A and D are merely downstream segments of vast watersheds, respectively, of the Columbia (which drains a similar area in Canada) and Snake Rivers.

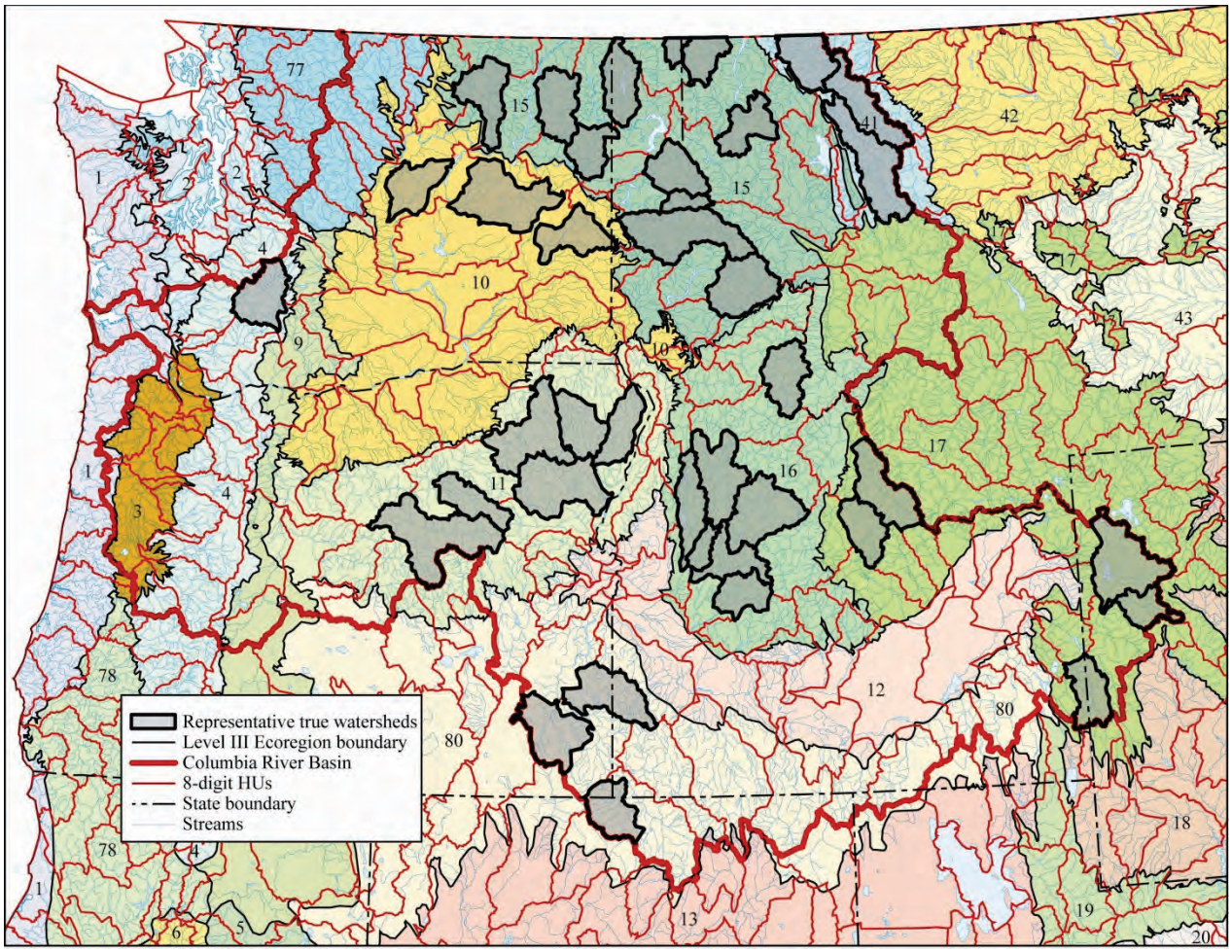

FIGURE 5-7

Representative 8-Digit HUs that are True Watersheds within Level III Ecoregions in the Columbia River Basin

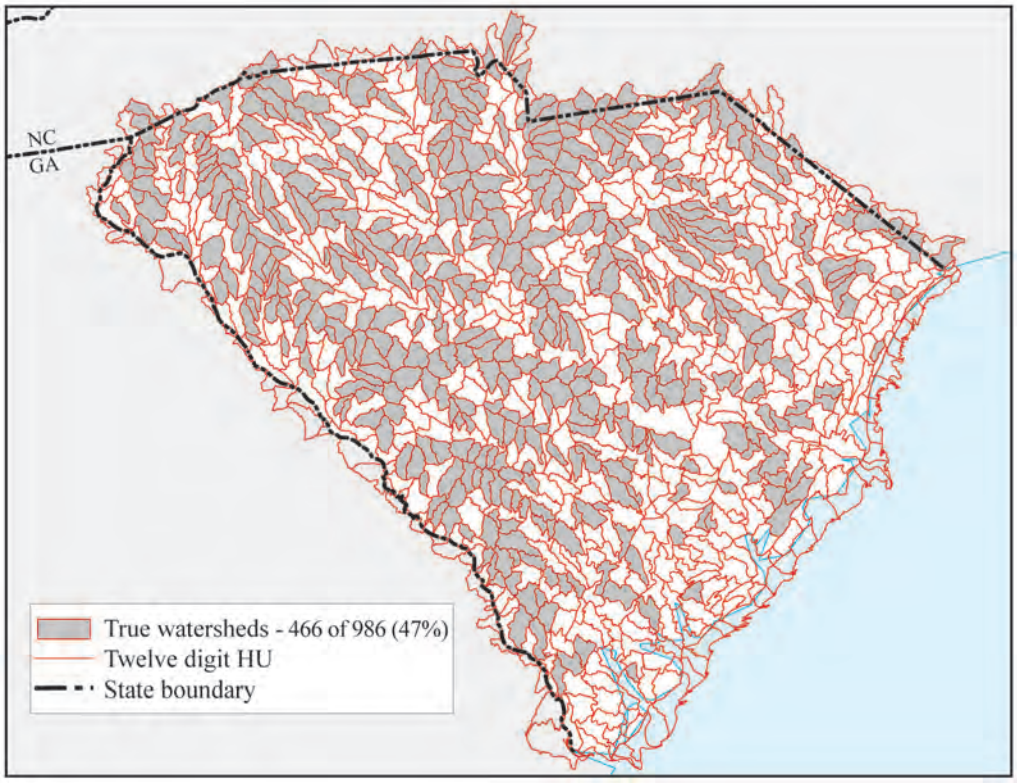

FIGURE 5-8

Twelve-Digit HUs in South Carolina that are True Watersheds. Note that only 47% are true watersheds.

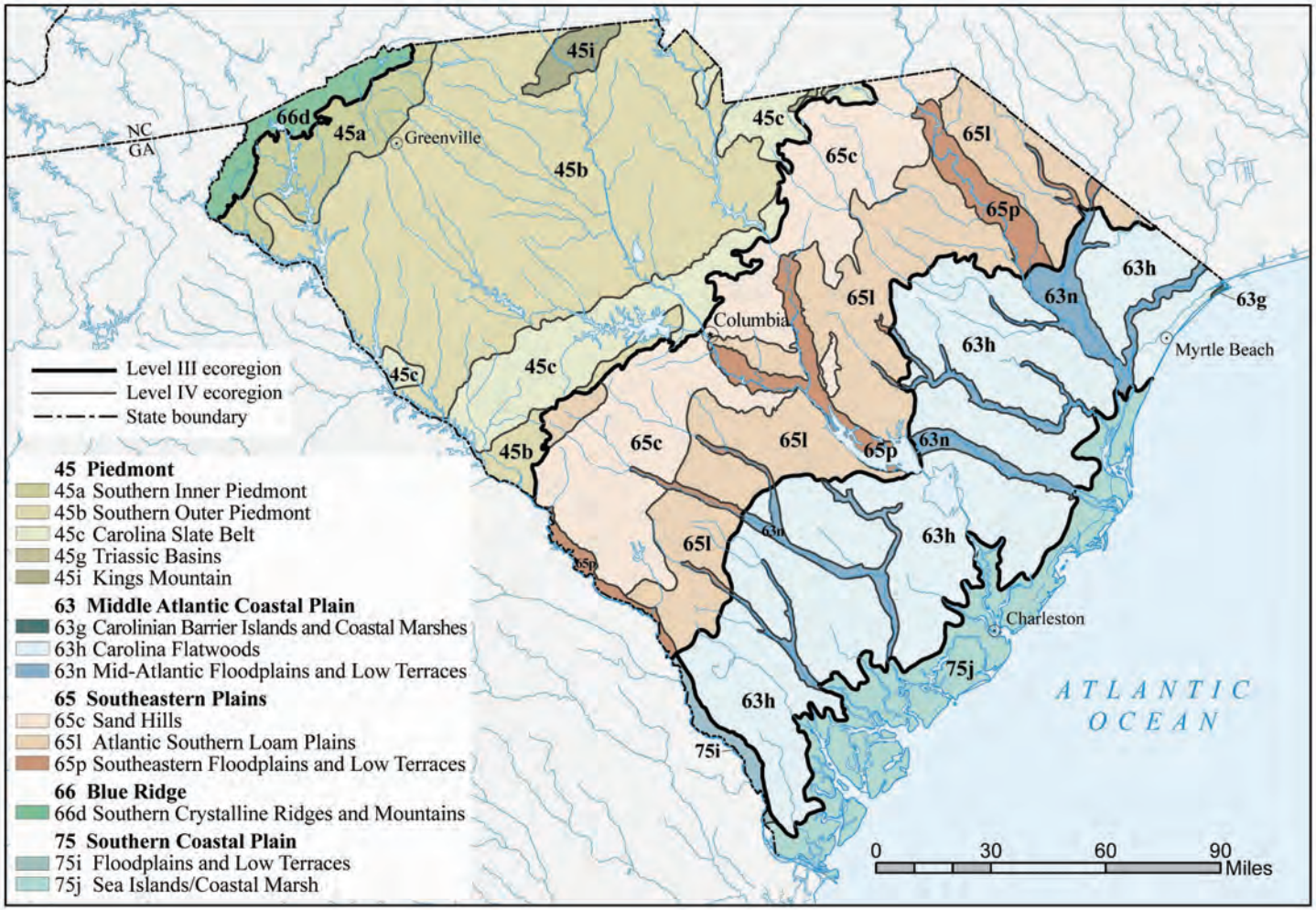

FIGURE 5-9

Level III and IV Ecoregions of South Carolina

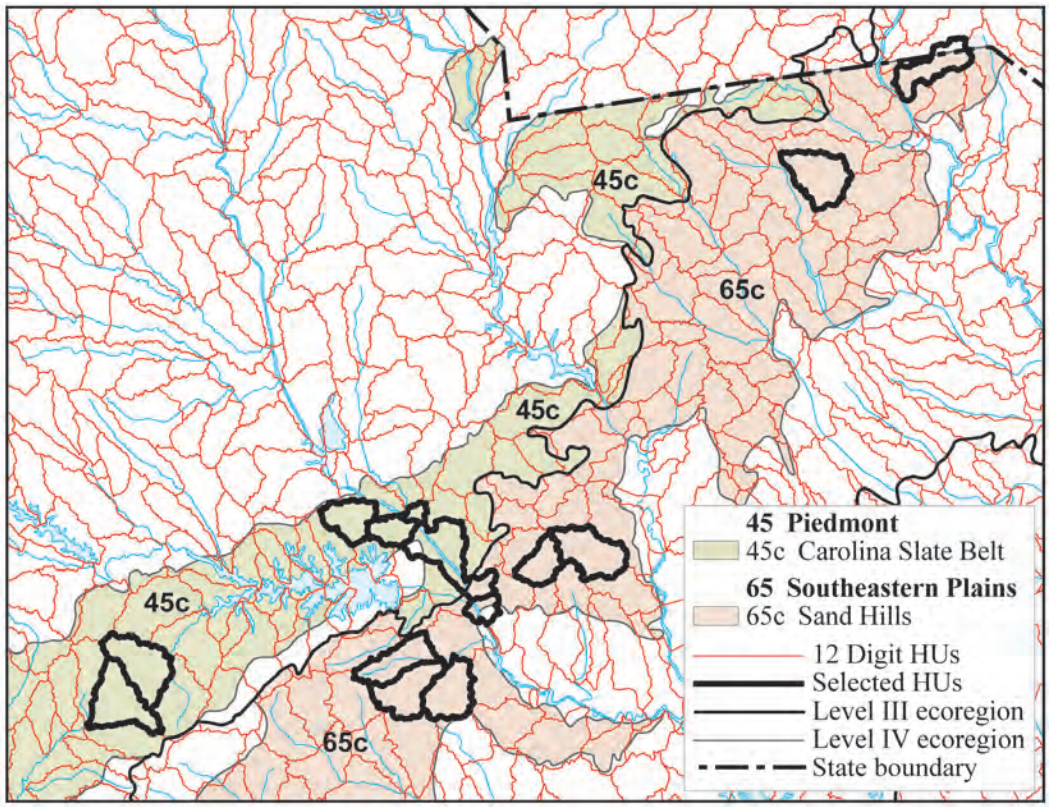

FIGURE 5-10

Selected 12-Digit HUs in the Carolina Slate Belt and Sand Hills Level IV Ecoregions of South Carolina

be very dissimilar to those with true watersheds in the other ecoregion. And 12-digit HUs with topographic watersheds extending far beyond the HU boundary are likely to be dissimilar to those that are true watersheds and are within a single ecoregion (see Figure 5-11). Consequently, the quality and quantity of water at the downstream points of the HUs that are not true watersheds (see A and B in Figure 5-11) will reflect the aggregate of all the characteristics in the different ecoregions they drain. Data from 12-digit HUs that are true watersheds completely or nearly completely within specific Level IV ecoregions (see Figure 5-12) are useful for determining patterns of water quality and flow regime in South Carolina. Variability among these types of reference watersheds will normally be less for those representing Level IV ecoregions than those of the larger Level III ecoregions. Data from aggregations of 12-digit HUs that comprise true watersheds that are completely within Level IV ecoregions (see Figure 5-13) are useful for determining water quality and flow regime characteristics in larger streams representative of those ecoregions.

A third limitation of HUs occurs when study sites are located other than at the downstream boundary of an HU or on a stream in an HU that includes multiple parallel and disconnected streams. When study sites are linked to such HUs in geographic analyses, much of the landscape is often downstream of the site or on the disconnected streams. The landscape conditions downstream of a site and on different streams have far less influence on the site than does the true watershed upstream of the site (Brendan et al., 2006; Hollenhorst et al., 2007).

Although not specifically an HU issue, there are two additional pitfalls in the use of the National Hydrography Dataset (NHD) deserving brief discussion: stream density and stream order. The NHD hydrography for South Carolina indicates that stream densities change abruptly at some 1:100,000 USGS topographic quadrangle boundaries (see Figure 5-14 and Table 5-1). In South Carolina, this change in densities is about double. This phenomenon occurs in many parts of the United States and can be traced to the differences in mapping specifications (quality control) among USGS mapping projects. The density differences can greatly affect sampling efforts and data interpretation in EPA’s regional and national stream assessments (Stoddard et al.,

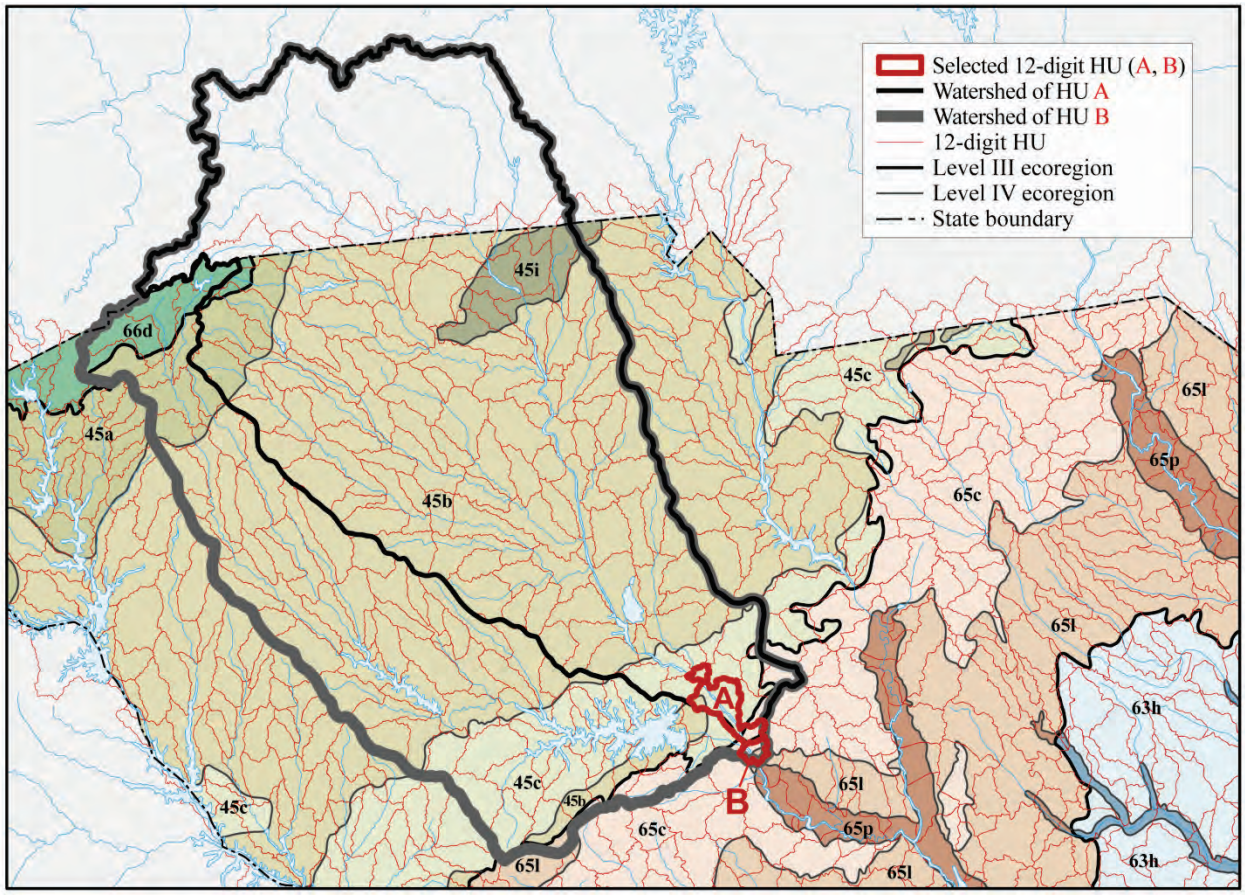

FIGURE 5-11

True Watersheds Associated with Downstream Points in Two of the Selected 12-Digit HUs Shown in Figure 5-10. Note that HU-A is a downstream segment of the Broad River watershed and HU-B is a downstream segment of an even larger area comprising over 200 12-Digit HUs making up the watershed of the Broad and Saluda Rivers, and draining three Level III ecoregions. The remaining selected HUs shown Figure 5-10 are true watersheds.

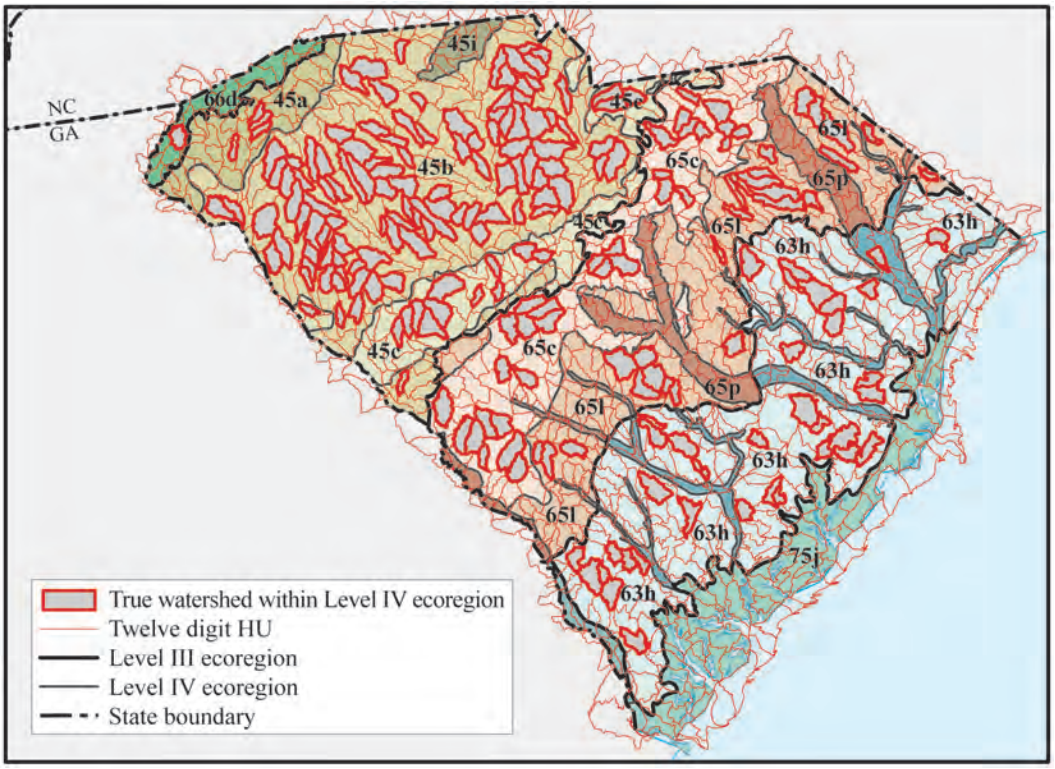

FIGURE 5-12

Examples of 12-Digit HUs that are True Watersheds Completely within, and thus Representative of, Specific Level IV Ecoregions of South Carolina

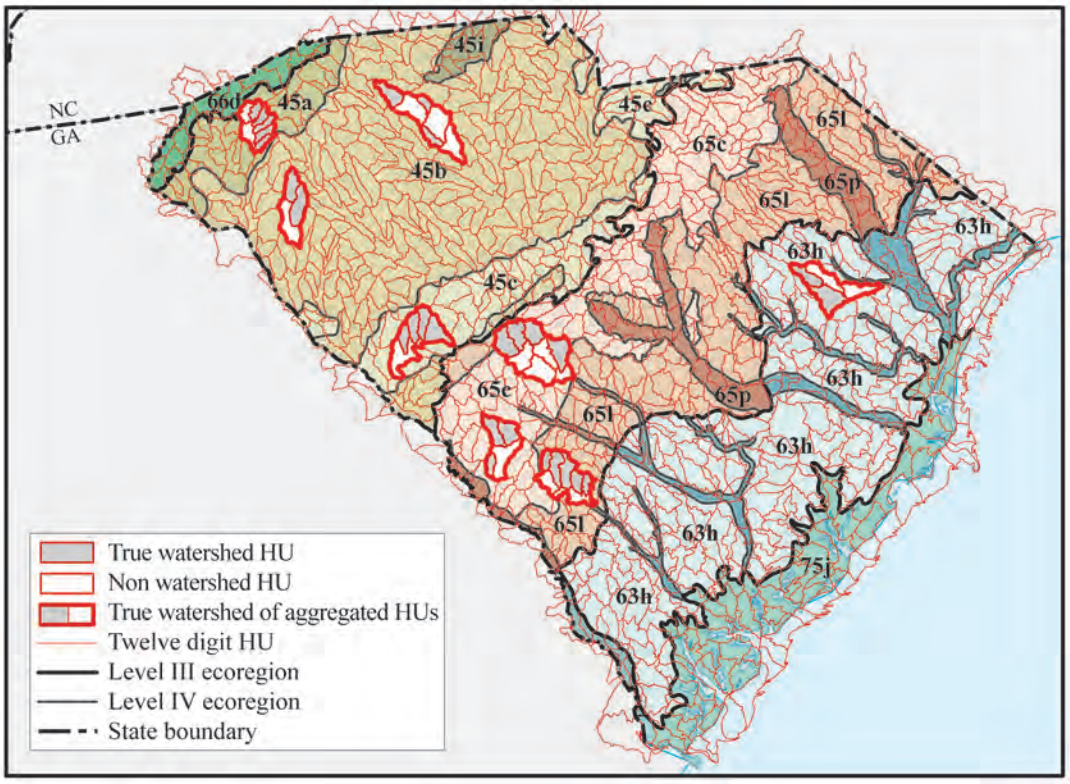

FIGURE 5-13

Examples of Aggregations of 12-Digit HUs that Together Comprise True Watersheds that are Completely or Nearly Completely within Specific Level IV Ecoregions of South Carolina

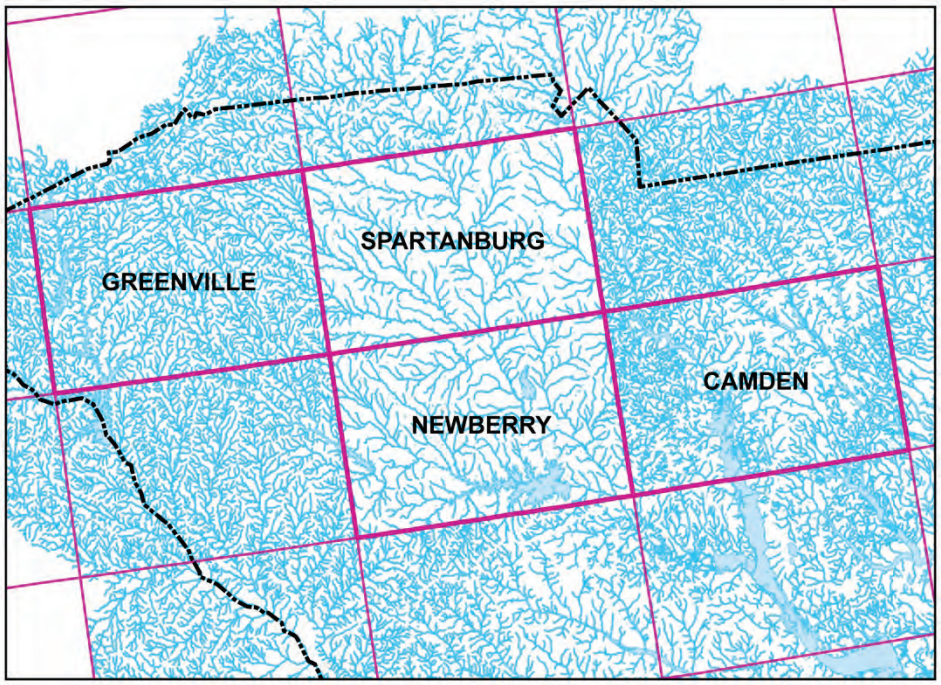

FIGURE 5-14

Stream Densities on 1:100,000 Scale USGS Topographic Maps in South Carolina. Note that the densities on the Greenville and Camden maps are about twice those on the Spartanburg and Newberry maps.

TABLE 5-1

Differences in NHD Flowline (rivers/streams and artificial path) Density between Four 1:100K Quads in South Carolina.

100K Quads	Total NHD Flowline (mi.)	NHD Flowline Density (mi/sq. mi.)
Greenville	2,468.43	1.26
Spartanburg	1,174.02	0.60
Newberry	1,357.81	0.69
Camden	2,634.83	1.34

2005; U.S. EPA, 2000, 2006), which selected sites on different order streams on the basis of NHD stream traces.

Not only does the NHD lead to errors in determining stream order, but stream order itself is a poor estimator of stream size or volume for several reasons. These include natural variation in the watershed area required to generate a channel and a permanent stream, inaccurate and imprecise field annotation, inconsistent mapping between xeric and humid regions, inconsistent map scales, and the degree to which ephemeral streams are included (Hughes and Omernik, 1981, 1983; Oberdorff et al., 1995; see Figure 5-15). In addition, stream order is confounded with influent streams, distributaries, spatially intermittent streams, streams flowing from other water or ice bodies, and streams in karst and glaciated regions. Instead of stream order, we recommend using a more accurate and meaningful indicator of stream size—such as estimated discharge (e.g., McGarvey and Hughes, 2008) or potential volume, estimated from the product of mean annual runoff and watershed area (Pont et al., 2009) or direct field measures of width or depth (Hughes et al., 2011). Pont et al. (2009) reported that potential stream volume was a significant predictor for vertebrate species richness, benthic species richness, and fish tolerance index in western U.S. streams. McGarvey and Hughes (2008) found a highly significant relationship between fish species richness and discharge for three large Oregon rivers. Further support for using a predictor of flow comes from Stanford et al. (1996) who argued that altered flows were the most pervasive threat to the world's rivers, and Poff et al. (1997) who deemed the flow regime as the master variable governing river character.

5.5. GENERAL PRINCIPLES

5.5.1. Computer Delineation of Ecoregions and Hydrologic Landscape Regions

There have been three noteworthy attempts to map ecological or hydrological regions through use of geographic information systems (GISs) and multivariate statistical analyses (Hargrove and Hoffman, 1999, 2004; Wolock et al., 2004; Sayre et al., 2009). The main rationale for these quantitative methods is that they are "more explicit, repeatable, transferable, and defensible than subjective models based on

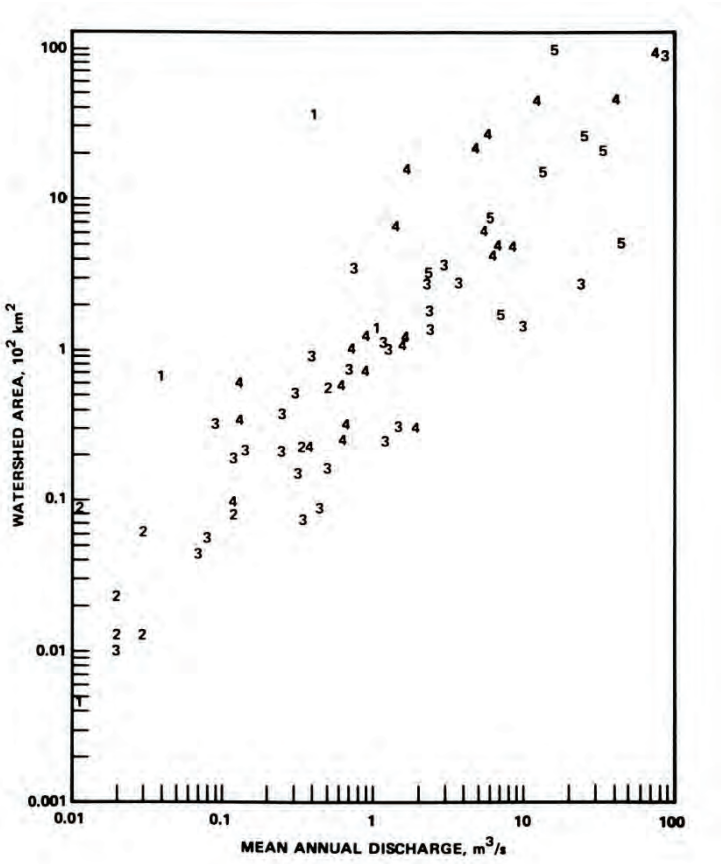

FIGURE 5-15

Watershed Areas and Mean Annual Discharges Relative to Stream Orders. Numbers are stream orders, each of which represents a study stream site in the United States with published results. Note that streams of any given order may vary by an order of magnitude in discharge or watershed area.

Source: Hughes and Omernik (1983).

human expertise" (Hargrove and Hoffman, 2004). Computer methods are explicit in that the developer/mapper can spell out which factors, source maps and data sets, and methods were used and how they were used. However, decisions concerning the characteristics to include and the data sets and maps used to portray those characteristics involve judgments by the mapper. Other scientists might disagree with those judgments and, if they were to start the same process from scratch without knowledge of the previous quantitative approach, would likely make very different judgments that would affect the final product. The use of qualitative, *weight of evidence*, methods (Omernik, 2004) for mapping ecological regions, on the other hand, recognizes that (1) the factors that are more or less important in giving each ecological region its identity, and how these factors are interrelated, vary from one region to another regardless of the level of detail (or hierarchical level) of mapping, and (2) the accuracy, level of generality, and relevance of the classification used in the maps representing each characteristic (e.g., soil, geology, physiography, and vegetation), vary from one map to another and from one area to another, even for maps published at the same scale (Omernik, 1995).

Omernik's experience in mapping military geographic regions reveals the importance of evaluating the accuracies, relevance of classifications, and levels of generality of all data sources including maps and numerical data (Omernik and Gallant, 1990). Understanding basic interrelationships and associations among geographic phenomena is key to filtering the data sources and ultimately mapping regions that fit the purpose of each map. Each mapper, if he or she understands these geographic interrelationships, will be able to adjust for these aspects and sketch areas where there is coincidence in combinations of characteristics that give each region its identity. Even if two or more individuals using this method are mapping an area independently, the resultant maps are usually similar. Here, the test of the regions is not in how they were compiled, but how useful they will be. Had only computer methods been employed, this filtering would have been difficult to impossible because of the infinite number of ways geographic characteristics are associated with one another and the many ways each can be represented on maps. Because human lives were at stake in mapping military geographic regions, it was vitally important to be generally right rather than precisely

and consistently wrong. The same should be true for mapping ecological regions because of the importance of maintaining and rehabilitating ecosystem condition.

Quantitatively developed maps of ecoregions or hydrologic regions often identify the same region in vastly different ecological settings. The map by Wolock et al. (2004) is made up of about 2,000 groups of HUs each of which is assigned one of 20 *hydrologic region* numbers. Each of these 20 regions comprises a particular combination of land-surface form, geologic texture, and climatic characteristics, but these noncontiguous hydrologic regions can be found in markedly different environmental conditions. For example, the map shows that one of the 20 regions occurs in southeastern Maine, the Cross Timbers region of Oklahoma, and south central Minnesota. Another region can be found in the Mississippi Alluvial Plain of southeastern Louisiana, the Lake Agassiz Plain in northern Minnesota and North Dakota, and the Subarctic Coastal Plain of southwest Alaska. Although such regions display hydrological similarities, their ecological similarities are minimal, and consequently it is named a hydrologic region map, not an ecoregion map.

Nonetheless, computer mapping exercises such as those of Hargrove and Hartman (2004), Wolock et al. (2004), and Sayre et al. (2009) are useful in locating areas with particular combinations of geographic characteristics and should prove useful in helping to explain the nature of ecoregions and ecoregion boundaries that have been developed more qualitatively. As has been previously noted, characteristics that distinguish ecological regions and the order of importance of the different characteristics vary from one region to another regardless of the hierarchical level of regionalization. Once these primary distinguishing characteristics have been determined, quantitative techniques can be employed to help illustrate how conditions vary within and between regions, particularly as these conditions relate to management issues and projected land use changes.

5.5.2. Strengths and Limitations of Using Watersheds and Ecoregions Together

Ecoregions and watersheds have very different purposes, but they can be complementary if used together correctly. An ecoregion provides a spatial framework in which the quantity and quality of environmental resources and ecosystems in general

can be expected to exhibit a particular pattern (Omernik and Bailey, 1997). It bears repeating that watersheds, where definable, are areas where water drains to a point. Compared with an ecoregion of similar size, a watershed tends to be more dissimilar in factors that influence water quality and quantity because watersheds tend to cross ecological regions. However, watersheds completely within an ecoregion will tend to be similar to one another and dissimilar to those in other ecoregions. Sets of these watersheds and their downstream points have been termed *reference watersheds*, and reference sites can be used to determine the central tendencies of ecoregions (Omernik, 1995) or the potential biotic and abiotic conditions of ecoregions (Hughes et al., 1986; Hughes, 1995; Bryce et al., 1999; Stoddard et al., 2005; Herlihy et al., 2008; Whittier et al., 2006, 2007). (Note the term reference is not used here as preferred condition but as existing condition). Comparisons of the natural and anthropogenic characteristics of such watersheds aid us in determining the factors associated with ecosystem differences in each ecoregion.

Watershed/Ecoregion Key Points

- Watersheds are imperative for understanding associations between human and nonhuman characteristics and water quality and quantity.
- Watersheds rarely correspond to areas within which there is similarity in characteristics affecting water quality and quantity.
- Most hydrologic units (HUs) are not watersheds.
- In some areas, watersheds are difficult to impossible to delineate or vary in their relevance.
- Watersheds and ecoregions are complementary frameworks.

Water quality of watersheds crossing more than one ecoregion will reflect characteristics of each of the ecoregions occupied. For example, watersheds completely within the Southeastern Plains of the Carolinas will be relatively similar to one another, as will watersheds completely within the Piedmont ecoregion in those states. Downstream points of watersheds that drain both of these ecoregions will reflect the characteristics of both ecoregions. Likewise, streams that originate in the Blue Ridge ecoregion and flow through the Piedmont, Southeastern Plains, and Atlantic Coastal Plain ecoregions will reflect contributions from all four of these regions. Water quality measured for ecoregion reference watersheds for each of these ecoregions will help in estimating these relative contributions (Omernik and Bailey, 1997).

Watersheds that are completely within each of the Level III ecoregions in the Columbia Basin (see Figure 5-7) can be used to determine the ecoregion effect on

points on those streams, such as for HU-D (see Figures 5-5 and 5-6). Obviously, ecoregions closer to the downstream point of HU-D will have a greater effect on the site than more remote ones such as ecoregion 17 (Middle Rockies). Much of the water draining from the Lost River and Teton Ranges (in ecoregion 17) is lost through evaporation and infiltration into the lava fields of the Snake River Plain or is used and reused for irrigation. However, if water were diverted outside the basin or if irrigation ceased, the quantity and quality of water measured at the downstream point of HU-D would be affected. Hence, when used together correctly, ecoregions and watersheds can provide a useful mechanism for meeting resource management goals outlined in the Clean Water Act, as well as for broader ecosystem management concerns of state and federal agencies, such as threatened and endangered fish species.

When using ecoregion reference sites or reference watersheds, it is important to recognize that these terms have many meanings (Stoddard et al., 2006). These meanings range from representing historic, natural or pristine condition (before human disturbance), to *best attainable* condition, to *realistically attainable* condition, to *least disturbed* condition, to *minimally impacted* condition, to *disturbed* condition (Stoddard et al., 2006; Bryce et al., 1999). There has been a tendency by some people to map ecological regions using naturally occurring characteristics and considering nature as if humans were not part of it. Such an approach is unrealistic. The same is true for expecting that it is possible to find stream sites or watersheds that reflect pristine conditions. It is now generally accepted that if humans were removed from North America, patterns in ecosystem components would not revert to those that existed before Europeans arrived or before Native Americans inhabited the continent. Too many plants and animals have been removed or introduced, and the land and water have been drastically altered through anthropogenic activities such as agriculture, urbanization, mining, and channelization (Omernik et al., 2000). In addition, there is considerable natural variation among least-disturbed reference sites (Whittier et al., 2006; Stoddard et al., 2008; Pont et al., 2009). However, least-disturbed reference conditions in ecoregions that are intensively used by humans are comparable to most-disturbed conditions in ecoregions that are less anthropogenically altered (Whittier et al., 2006; Pont et al., 2009). Even in the latter, there are clear disturbance gradients

indicating ample room for marked improvement in ecological condition (Hughes et al., 2004; Pont et al., 2009).

It is unlikely that any single predictor variable, especially a class variable, will explain any continuous and complex response variable, such as an ecosystem or assemblage. In some recent research, watersheds and ecoregions have been found to explain less biological variability than local, quantitative environmental measures or GIS variables (Hughes et al., 2006). For example, researchers have used such data as stream depth, velocity and conductivity to predict the assemblage condition of aquatic macroinvertebrates (Moya et al., 2007) or their taxa richness (Clarke et al., 1996; Reynoldson et al., 1997; Davies et al., 2000; Hawkins et al., 2000b; Bailey et al., 2004). Others have used GIS data (geology, elevation, slope, mean annual air temperature, catchment area, runoff) for predicting the richness of fish assemblages (Joy and De'ath, 2002) or their assemblage condition (Oberdorff et al., 2001, 2002; Tejerina-Garro et al., 2006; Pont et al., 2006, 2009). Also see Snelder and Biggs (2002) and Seelbach et al. (2006) for weight-of-evidence classification of rivers based on catchment, valley, and channel characteristics. Cluster analysis of habitat data, biological data, or both is also being used in conservation planning as more digital data become available (Belbin, 1993; Ferrier et al., 2002, 2007; Snelder et al., 2007). In addition, basins and ecoregions were reported to account for no more site-scale assemblage variability than political or null polygons (Hawkins et al., 2000a; Van Sickle and Hughes, 2000; McCormick et al., 2000; Waite et al., 2000; Herlihy et al., 2006). Finally, both HUs and ecoregions include markedly different size rivers, and river size is a major determinant of ecosystem character. For this reason, Pflieger (1971) classified the great rivers of Missouri as separate regions, but whatever their locations, expectations for many ecological variables must be calibrated against river size (Fausch et al., 1984; McGarvey and Hughes, 2008; Pont et al., 2009). We therefore reemphasize that ecoregions should not be used alone in quantitative prediction of individual aquatic ecosystem components, but in combination with other quantitative and continuous environmental data if they are available.

Nonetheless, when used together correctly, ecoregions and watersheds are useful for designing research and monitoring programs and for interpreting biological

patterns over large areas (Hocutt and Wiley, 1986; Hughes et al., 1987, 1994; Stoddard et al., 2005; U.S. EPA, 2000, 2006; Frimpong and Angermeier, 2010) or where natural gradients are strong (e.g., mountains versus plains, Omernik and Griffith, 1991; Pinto et al., 2009; Moya, 2011). We, therefore, suggest using ecoregions and watersheds to do the following:

- Predict ecosystem conditions when other data or models are unavailable.
- Predict stressors and likely stressor-response patterns.
- Examine patterns in, and reporting aspects of, ecosystem services at different scales nationally and regionally.
- Evaluate patterns in and threats to both terrestrial and aquatic biodiversity.
- Select and calibrate reference sites.
- Interpret patterns in ecological data.
- Report patterns in ecological condition.
- Forecast where climate change effects will likely be greatest.
- Plan management strategies and tactics.

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