Nationwide Assessment of Nonpoint Source Threats to Water Quality

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Water quality is a continuing national concern, in part because the containment of pollution from nonpoint (diffuse) sources remains a challenge. We examine the spatial distribution of nonpoint-source threats to water quality. On the basis of comprehensive data sets for a series of watershed stressors, the relative risk of water-quality impairment was estimated for the over 15,000 fifth-level watersheds in the contiguous United States. A broad division emerged at about the 100th meridian, with eastern areas typically under higher stress than western areas, reflecting the generally higher housing, road, and agriculture densities and higher levels of atmospheric deposition in the eastern division. Recent trends in some stressors are encouraging, but the prospect of further substantial population growth indicates continued pressure on water quality, suggesting that renewed focus on controlling nonpoint-source pollution will be needed if the goals of the Clean Water Act are to be attained.

Keywords: pollutants, environmental health, assessments

Since passage of the Clean Water Act in 1972, considerable progress has been made in controlling the pollution of the nation's freshwaters (Smith et al. 1987, Lettenmaier et al. 1991). However, most of the successes have been with point sources of water pollution, such as factories and municipal wastewater-treatment plants. Nonpoint sources, such as farms, roadways, and urban or suburban landscapes, remain largely uncontrolled. Recent US Environmental Protection Agency national water-quality inventories showed that five of the top six identified water-quality-related sources of river and stream impairment in the United States were nonpoint sources (USEPA 2009, 2011a). To take a specific example concerning nutrients, in a recent national summary, Dubrovsky and Hamilton (2010) found "limited national progress… in reducing the impacts of nonpoint sources of nutrients" (p. 12).

Given the continuing concern about nonpoint-source pollution, we sought to understand how the threats to water quality from nonpoint sources vary across the nation. This approach indicates the risk of impaired water quality, and by *risk* we mean the relative possibility rather than an estimate of probability. We assess this risk for the over 15,000 fifth-level (10-digit) watersheds in the contiguous United States (NRCS 2011), providing a comprehensive picture of the relative risk of water-quality impairment. This is done for a parsimonious set of watershed stressors that are known to affect one or more of three common water-quality problems: sediment, nutrients, and toxics.

Combining across the stressors, the watersheds are ordered from the lowest to the highest level of concern. Such an ordering has been called a *disturbance index* (e.g., Stein et al. 2002, Wang et al. 2008, Falcone et al. 2010) because it measures the extent to which the landscape has been altered by human activities. However, unlike some other recent assessments (e.g., Paulson et al. 2008, Wang et al. 2008), we make no attempt to establish a reference condition against which the watersheds can be compared. Restricting the assessment to an ordering of watersheds places no value judgments on the condition of any one watershed but does provide clear relative information, showing where the watersheds with the greatest likelihood of impaired water quality are found.

Although our assessment is ambitious in scope—including over 15,000 watersheds—it is modest in depth and intended to offer only a broad spatial comparison of the risk of water-quality impairment. Furthermore, note that the mere presence of a stressor is only a first-order indication of water-quality problems. The actual level of risk depends on many site-specific characteristics, including background conditions such as slope or soil type and human actions such as the extent to which best-management practices such as conservation tilling or upgrading culverts have been implemented. Without detailed information on such conditions and actions, the assessment does not provide a basis for site-specific mitigation actions. Nevertheless, the assessment should be useful, both to focus more in-depth study and to provide a broad-scale understanding of the risk of nonpoint-source water-quality problems posed by human actions across the national landscape.

Measuring relative risk

We began with the complete set of fifth-level watershed boundaries posted by the Natural Resources Conservation Service in March 2009 (NRCS 2009). Excluding Alaska and Hawaii because of missing data on some stressors and a few watersheds consisting largely of water (mainly along the coasts) left 15,272 watersheds for analysis, which range in size from 35 to 8319 square kilometers (km^2) and have a median area of 467 km^2, with 97% being from 100 to 1000 km^2 in size.

To provide a broad spatial comparison, findings are summarized by water-resource region (WRR). WRRs, of which there are 18 in the coterminous United States (figure 1), are very large river basins or collections of large coastal watersheds (USGS 2009). They contain between 310 and 2429 watersheds and vary widely in vegetative cover, land ownership, and other characteristics (table 1).

Many water-quality problems could be studied, including sediment, nutrients, dissolved metals and other toxic chemicals (which together we will call *toxics*), the alteration of stream temperature, and pathogens. Lacking good-quality nationwide data sets for variables relevant to some of these problems, we limited our effort to sediments, nutrients, and toxics, all of which can end up in lakes and streams, thereby potentially damaging valuable resources. These three problems encompass 6 of the 10 most frequently cited causes of fresh-water impairment in US rivers and streams; the others are flow alternation, habitat alternation, water-temperature change, and pathogens (USEPA 2011a).

Stressors

Measures of watershed stressors vary from the coarse (e.g., watershed road density) to the more refined and site specific (e.g., the density of roads in riparian areas). The refined measures allow a more-precise assessment of risk but require much additional data processing (e.g., to delineate riparian areas). The use of more-refined measures is not uncommon (Wang et al. 2008, Brown and Froemke 2010). However, Falcone and colleagues (2010) recently used independent measures of watershed degradation to assess the utility of different sets of stressors and found that the refined measures do not tend to contribute much to an overall assessment of risk, in part because of the high correlations between the coarse and the corresponding refined measures. In light of this finding and of the difficulty of measuring refined stressors for the entire United States, we restricted this assessment to more coarse measures.

Each selected water-quality problem—sediment, nutrients, and toxics—is the result of a series of stressors. We used nine stressors to characterize the risk of the three problems (table 2). Our approach emphasizes human-caused stressors, with the exception of wildfire. Wildfire is a special case, because although wildfire is a natural phenomenon, the severity of wildfire has been enhanced by past management actions in many forests with short fire-recurrence intervals (Swetnam et al. 1999). For this reason, potentially damaging wildfire in selected vegetation types was included as a stressor. The following paragraphs briefly describe the three problems and review literature supporting the selection of the stressors.

Sediments are soil particles that are carried along in streamflow, some of which settle on stream bottoms or in lakes, reservoirs, canals, and pipes. Suspended sediments increase turbidity and transport attached nutrients, toxics, pathogens, and other potential pollutants and can reduce photosynthesis by algae, can reduce the success of sight-feeding fish, and can degrade the quality of drinking water (or increase the cost of water treatment). Excess amounts of settling particles reduce the porosity of gravel beds used by spawning fish, lower the storage capacity of lakes and reservoirs, clog water-diversion structures, and interfere with navigation.

Sediment levels vary as a function of land cover and are affected by land management. On the basis of 1982 National Resource Inventory estimates of erosion on nonfederal rural lands, in light of sediment transport and delivery predictions, the sediment-loading rates into rivers and streams from cropland were estimated to be more than five times the rate from forest land (Gianessi et al. 1986). Jones and colleagues (2001), in studying the

Figure 1. Water resource regions.

Table 1. Watersheds analyzed.

Water resource region	Number of watersheds	Total area of watersheds (1000 km²)	Percentage of watershed area[a]					Precipitation (mm per year)[a]	Mean elevation (m)
			Federal land	Forest	Range land	Water or wetland	Cultivation		
1. New England	373	155	4	68	5	12	5	1154	810
2. Mid-Atlantic	646	265	6	56	1	7	24	1106	938
3. South Atlantic Gulf	1515	687	8	42	10	20	19	1340	343
4. Great Lakes	653	302	12	36	5	17	31	873	951
5. Ohio	1010	422	8	49	3	2	37	1137	1063
6. Tennessee	217	106	22	58	5	4	24	1407	1412
7. Upper Mississippi	1155	492	4	21	3	8	59	864	963
8. Lower Mississippi	509	261	6	25	6	25	36	1436	218
9. Souris Red Rainy	310	153	15	15	7	19	55	539	1318
10. Missouri	2429	1324	21	9	52	3	32	540	3261
11. Arkansas White Red	994	642	9	21	43	3	28	813	2297
12. Texas Gulf	670	465	3	14	48	7	24	822	1192
13. Rio Grande	550	343	33	11	84	1	2	385	4900
14. Upper Colorado	523	294	77	29	61	2	3	389	7047
15. Lower Colorado	548	363	71	18	76	1	2	332	4303
16. Great Basin	650	361	80	16	73	1	3	319	5806
17. Pacific Northwest	1526	709	59	40	43	2	10	861	3951
18. California	994	414	52	25	52	2	10	598	2888
Total	15,272	7758	27	27	37	7	23	790	2575

km², square kilometers; m, meters; mm, millimeters.
[a]For measurement details, see supplemental table S4.

Table 2. Stressors.

Stressor	Measure[a]	Problem		
		Sediments	Nutrients	Toxics
Housing density	Housing units per km² in year 2000	X	X	X
Road density	Meters of road and railroad per km² of watershed land	X		X
Cultivation	Percent of watershed area in agricultural land cover	X	X	X
Livestock grazing	Animal units per km² in year 2007	X	X	
Confined animal feeding	Animal units per km² in year 2007		X	X
Mining land cover	Percentage of watershed in mining land cover	X		
Potentially toxic mines	Total number of active and inactive mine sites potentially yielding toxics per 1000 km² of watershed			X
Potentially damaging wildfire	Percentage of area with a high risk of losing key ecosystem components in a forest fire	X		
Atmospheric deposition	Mean annual (2000–2006) deposition (in kilograms per hectare) of NO_3^- and SO_4^{2-} in wet atmospheric deposition		X	X

km², square kilometers; NO_3^-, nitrates; SO_4^{2-}, sulfates.
[a]For measurement details, see supplemental table S4, available online at *http://dx.doi.org/10.1525/bio.2012.62.2.7*.

relatively urbanized Mid-Atlantic region, found that stream-suspended sediment loads were positively correlated with the percentage of the watershed in urban cover and negatively correlated with wetland and riparian forest covers. In addition to the activities associated with the land covers of concern—plowing and other soil disturbances with agriculture (National Research Council 2010), construction in populated areas, and mining—three other activities or conditions are recognized as important causes of suspended sediment: road construction and subsequent road presence, which both exposes soil surfaces and concentrates surface runoff, thereby increasing sediment-transport capacity (Forman and Alexander 1998); severe forest fires (e.g., Benavides-Solorio and MacDonald 2001, Cannon et al. 2001, Moody and Martin 2001, Neary et al. 2005); and livestock grazing, especially in riparian areas (Kauffman and Krueger 1984, Wohl and Carline 1996, Belsky et al. 1999). On the basis of

this evidence, increases in the amounts of suspended sediment are likely to be associated with the following stressors: agriculture, housing (as an indication of urban, suburban, and rural domestic land uses), roads, mining, livestock grazing, and severe wildfire (table 2).

Nutrients—mainly forms of nitrogen and phosphorous—are essential for primary production in aquatic ecosystems, but at high levels, they can lead to excessive algal growth, which can cause murky water; can deplete the amount of dissolved oxygen, which is needed by fish and other aquatic organisms; and can alter aquatic species composition. Coastal eutrophication caused by excess nutrients can result in blooms of harmful or toxic algal species, a loss of sea grass beds and other important estuarine habitats, and changes in marine species composition, among other impacts (Driscoll et al. 2003).

Background concentrations of nutrients tend to be low in streams draining areas of natural land cover that are not subject to atmospheric nitrogen deposition, but land uses and atmospheric deposition can cause concentrations to rise far above those background levels (Mueller and Helsel 1996, Dubrovsky et al. 2010). Nonpoint-source loadings of nutrients are estimated to be over five times as great as point-source loadings (Carpenter et al. 1998). Annual nutrient concentrations in US streams draining predominately agricultural watersheds were found to be about nine times higher than those in streams draining predominantly forested watersheds and about four times higher than those in streams draining predominantly rangeland watersheds (Omernik 1977). Agricultural fertilizer, livestock manure (from both grazing and feedlots), and atmospheric deposition have consistently been found to be primary nonpoint sources of nutrients in streams (Smith et al. 1987, Carpenter et al. 1998, Mallin 2000, Jones et al. 2001, Driscoll et al. 2003, Dubrovsky et al. 2010, National Research Council 2010). Housing areas are additional nonpoint sources, most importantly because of landscape fertilizers, pet wastes, and septic leachate (Carpenter et al. 1998, Driscoll et al. 2003, Dubrovsky et al. 2010). On the basis of this evidence, the following stressors are likely to be important for nutrients: agriculture, housing, grazing and feedlots, and atmospheric nitrogen deposition (table 2).

Toxics are chemicals that cause damage to plants and animals (both invertebrates and vertebrates and including humans) at low levels. They include toxic heavy metals (e.g., mercury, lead, cadmium, arsenic), some pesticides (including herbicides), some industrial chemicals (e.g., PCBs [polychlorinated biphenyl]), some pharmaceuticals, and some acids.

Heavy metals exist naturally and are essential for life but are toxic at elevated concentrations. Mining can greatly increase heavy-metal loadings, and long-abandoned mining sites may continue to contribute such metals to the stream as metal-bearing fine sediments are washed out of tailings by heavy rains or enter the surface water from polluted shallow groundwater near the mines (e.g., Roline 1988, Rösner

1998, Courtney and Clements 2002). Other sources of toxic levels of heavy metals include vehicle traffic (e.g., Albasel and Cottenie 1985, Forman and Alexander 1998) and thermoelectric-power-plant emissions. Pesticides are used primarily on agricultural lands and in urban and suburban areas. Pesticide application rates on agricultural land tend to be roughly 1000 times greater than the rates on forested land (Brown and Binkley 1994). Although the level of herbicide application in urban areas does not approach that in agriculture, the uses of insecticides have been found to be similar (e.g., Hoffman et al. 2000, Paul and Meyer 2001). Nonpoint sources of pharmaceuticals are mainly associated with the presence of domesticated animals. Acids (primarily sulfuric and nitric acid) reach the soil or water bodies in acidic precipitation resulting from the burning of fossil fuels (e.g., at power plants, in vehicles). These acids can lower the pH of the water bodies into which they fall. Furthermore, when washed into the soil, the acids, if they are not buffered sufficiently, can leach toxic substances from the soil into receiving waters, possibly causing fish mortality or reproductive failure (Driscoll et al. 2003). Although the acidity of many water bodies (especially in the Northeast) has decreased in recent decades (Lettenmaier et al. 1991, Clow and Mast 1999, Stoddard et al. 1999, Burns et al. 2006), mainly in response to decreases in the deposition of sulfates, large differences remain across regions of the United States (see supplemental tables S1 and S2, available online at *http://dx.doi.org/10.1525/bio.2012.62.2.7*), and acid deposition remains a concern in many areas. On the basis of this evidence, the following nonpoint-source stressors are relevant to toxics: the mining of heavy metals and other toxic substances, agriculture, human activities associated with housing and landscape maintenance, vehicle traffic, feedlots, and acidic atmospheric deposition (table 2).

Scaling and weighting

The stressor variables must be combined to reach an overall measure of the risk of impaired water quality. Ideally, a multivariate model would be available that, across all target ecosystems, accounted for all relationships between stressors and water quality. Because watersheds are so complex and varied and onsite inventory is expensive, such a comprehensive model will be an unmet goal for many years to come, especially across the vast array of landscapes at issue here.

The procedure that we employ for combining across stressors is similar to that used in other recent assessments of environmental risks (Jones et al. 1997, Stein et al. 2002, Wang et al. 2008, Falcone et al. 2010, Vörösmarty et al. 2010). It relies on scaling to convert the stressor values to a common unit of measurement, avoidance of stressors that are highly intercorrelated (see supplemental table S3 for the intercorrelations among stressors), weighting that reflects the relative importance of the different stressors, and summation of the weighted estimates. The fundamental assumptions of this weighted-sum procedure are that the relation between each stressor and risk is linear and that the

effects of the stressors are additive (i.e., that the stressors do not interact). Although the assumptions are most probably violated to some extent, at least in some locations, they are unavoidable—because of the lack of a quantified measure of risk that could be used to model the relations of risk to the stressors—and are therefore commonly made. Careful attention to functional form and interactions, of course, is more feasible in localized studies.

Scaling is commonly performed using normalization, where the variables are each transformed linearly to a scale with a common range, such as a scale ranging from 0 to 1. Falcone and colleagues (2010), however, compared this procedure to one in which the values for each stressor were converted to percentiles and found that the percentile approach resulted in a more accurate ranking of watersheds (their data allowed an independent measure of impairment that could be compared with the stressor-based measure). If the stressor variables are positively skewed, using the percentile approach as opposed to normalization deemphasizes the importance of high stressor values and provides enhanced discrimination among sites with lower values. The distributions of most of the watershed stressors used here are highly skewed; compared with a skewness coefficient of 0 for a symmetrical distribution, the skewness coefficients of the present nine variables vary from 0.5 to 18.4, with all but one being above 1.3 and six being above 4.0, which indicates that the cases are clustered to the left of the mean, with most extreme values to the right. We employ the percentile approach using five categories such that each stressor variable is transformed to a scale ranging from 1 (lowest 20% of the values) to 5 (highest 20% of the values).

The percentile scaling and weighted-sum procedure was followed for each problem (sediment, nutrients, toxics) using weights based on the results of a principal component analysis (PCA; table 3). (For comparison, a set of equal weights was also used, as is reported in the Results section.) Establishing weights using PCA emphasizes the amount of variability in the data that is explained by the individual stressors and is commonly used in such assessments (e.g., Danz et al. 2007, Seilheimer et al. 2009, Primpas et al. 2010). The PCA procedure that we used is similar to that described by Falcone and colleagues (2010), who compared approaches and found the PCA approach to be slightly superior to the alternatives. Using the data from all of the watersheds, the procedure was followed separately for each problem as follows: (a) a PCA was performed (using the unrotated correlation matrix) on the stressors selected for the problem (see table 2), including all components with an eigenvalue greater than 0.85; (b) for each component, the loadings of the stressors were standardized to range from 0 to 1 and then multiplied by the eigenvalue of that component, thereby giving greater weight to stressors loading heavily on components that explain the most variance in the data; (c) for each stressor, the weighted loadings were summed across components; and (d) the sums for the stressors were linearly transformed to add up to 1 (table 3).

Table 3. Stressor weights by problem.			
Stressor	Sediment	Nutrients	Toxics
Housing density	.18	.13	.15
Road density	.20		.18
Cultivation	.21	.24	.20
Livestock grazing	.18	.20	
Confined animal feeding		.20	.19
Mining land cover	.14		
Potentially toxic mines			.09
Potentially damaging wildfire	.09		
Atmospheric deposition		.24	.19

Although the purpose of the PCA was to weight the stressors, it is instructive to observe the components that emerged (eigenvalue > 0.85), which are listed here in order of decreasing importance (i.e., decreasing percentage of variance explained). For the sediment problem, four components emerged: Component 1 loaded strongly to built features (housing and roads), component 2 to agriculture (cultivation and grazing), component 3 to mining land cover, and component 4 to wildfire. For nutrients, two components emerged: Component 1 loaded to agriculture (cultivation, animal feeding, and grazing) and component 2 loaded to housing. For toxics, three components emerged: Component 1 loaded to built features (housing and roads), component 2 to agriculture (cultivation and feeding), and component 3 to potentially toxic mines.

The weighted-sum procedure yielded a measure of relative risk, called a *risk value*, for each problem. To produce an overall risk value across the problems, the weighted-sum procedure was again used to combine across problems, with equal weights assigned to the problems. Finally, for presentation, the risk values were converted to a categorical scale of five risk levels, each representing one-fifth of the risk value range (1 indicates low risk, and 5 indicates high risk).

Results: Stress and risk levels

The stress levels varied greatly among the watersheds. For example, housing densities ranged from 1 to nearly 1900 units per km^2, livestock grazing densities ranged from 0 to over 400 animal units per km^2, and the density of toxic mines varied from 0 to 1.8 mines per km^2. For most stressors, stress levels tended to be highest in the eastern half of the United States. This phenomenon was most evident for housing density, road density, cultivation, livestock grazing, animal feeding, and atmospheric deposition (figure 2).

Interestingly, the average stress levels of the much larger WRRs also varied substantially, indicating a large-scale heterogeneity in stress levels across the United States. The highest stress levels were generally found in the eastern half of the country (WRRs 1–9). For example, at the WRR level, the minimum levels of all but one stressor (damaging wildfire) were found in the western half of the country (WRRs

10–18), and the maximum levels of all but two stressors (density of livestock grazing and the number of potentially toxic mines) were found in the eastern half (for WRR averages, see table S1).

Combining across the three problems, figure 3 shows the overall risk levels of the 15,272 watersheds. Most striking in the figure is the division between the eastern and western portions of the country, with the dividing line falling roughly
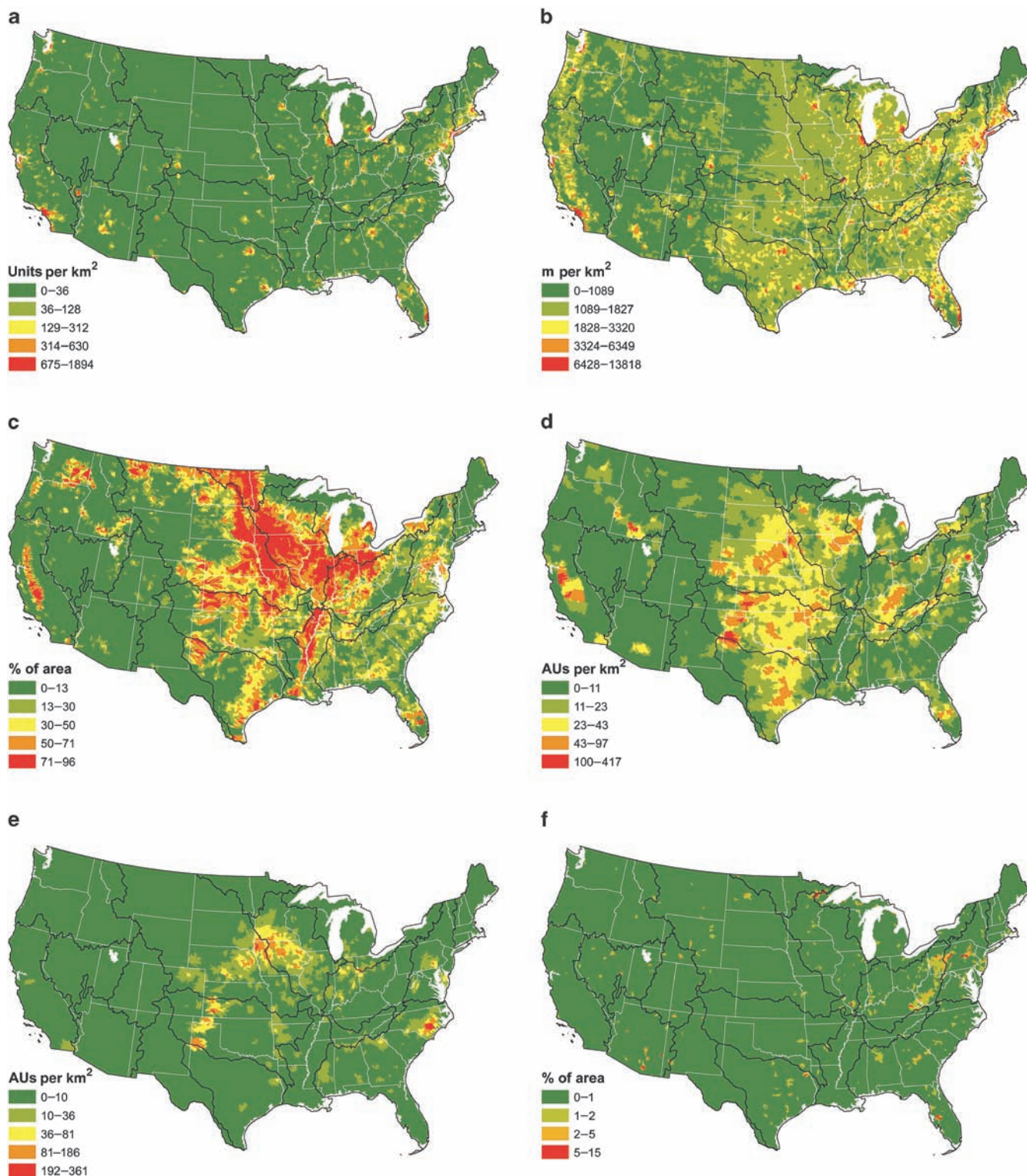
Figure 2. Levels of the stressors: (a) housing density, (b) road density, (c) cultivation, (d) livestock grazing, (e) confined animal feeding, (f) mining land cover, (g) the number of potentially toxic mines, (h) potentially damaging wildfire, (i) atmospheric deposition. Abbreviations: %, percentage; AU, animal units; ha, hectares; kg, kilograms; km², square kilometers; m, meters.
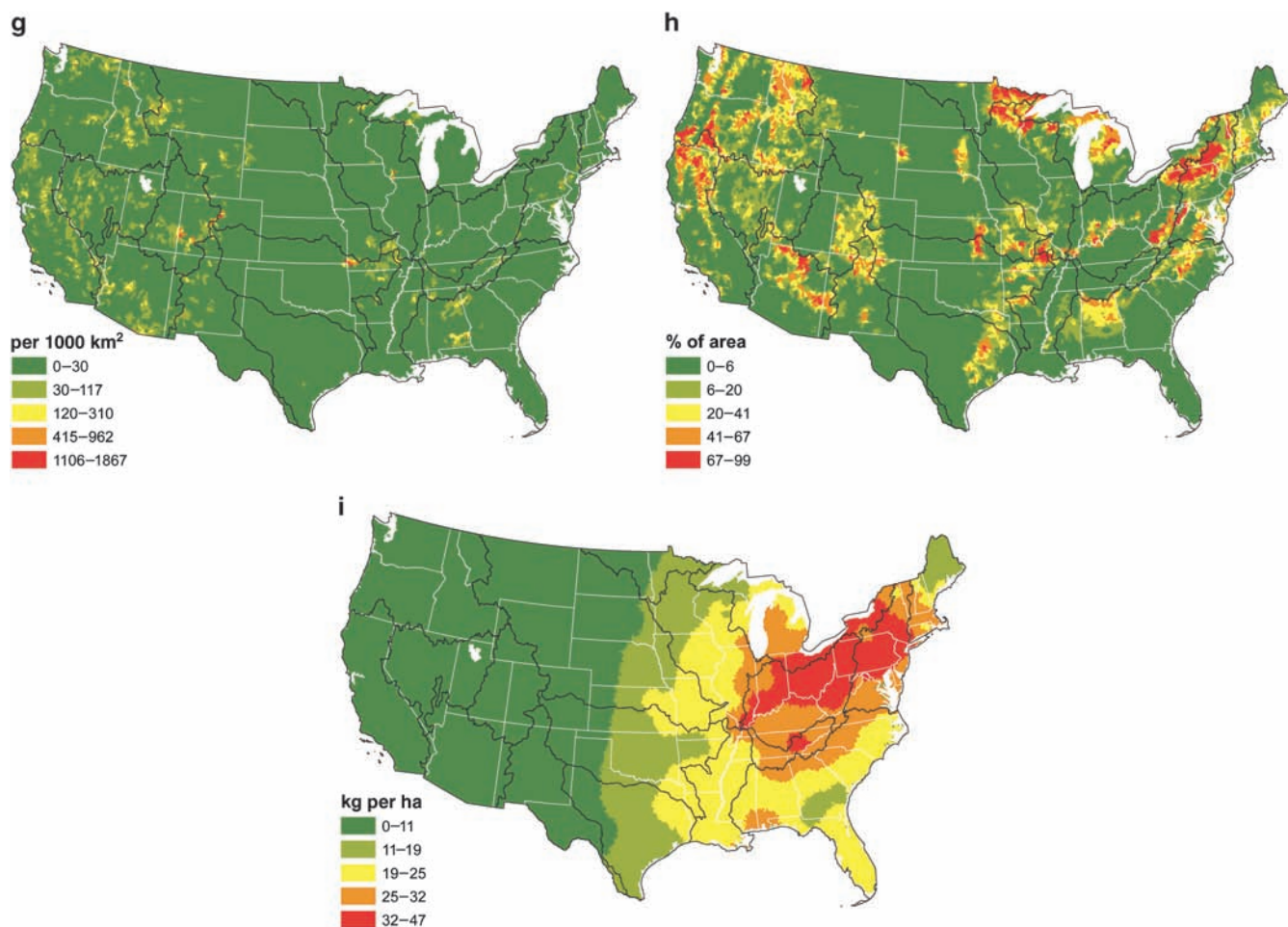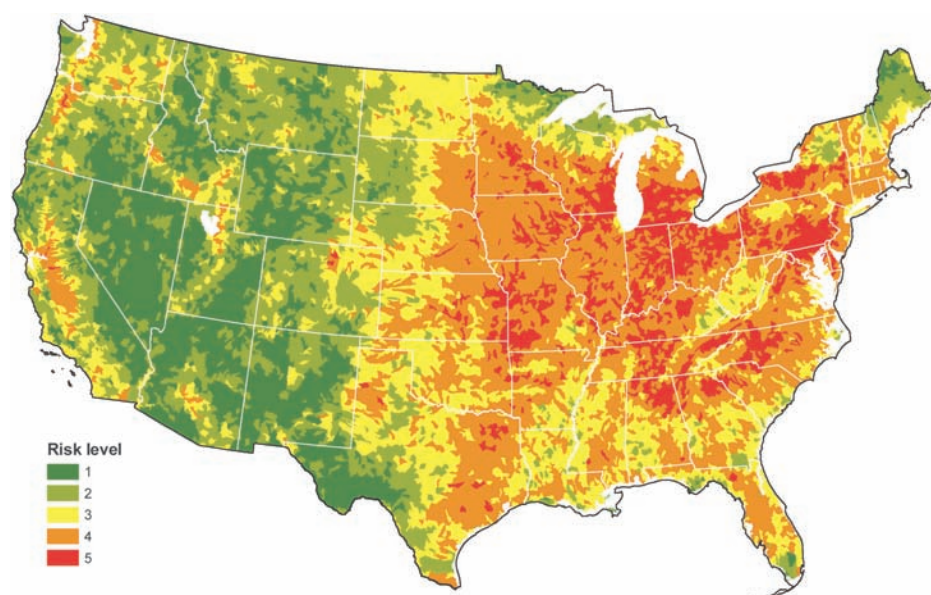
g

per 1000 km²
- 0–30
- 30–117
- 120–310
- 415–962
- 1106–1867

h

% of area
- 0–6
- 6–20
- 20–41
- 41–67
- 67–99

i

kg per ha
- 0–11
- 11–19
- 19–25
- 25–32
- 32–47

Figure 2. (Continued)

Risk level
- 1
- 2
- 3
- 4
- 5

Figure 3. Overall risk of water-quality impairment for 15,272 watersheds.

western areas, although there are notable exceptions. In the East, the areas of exceptionally low risk include much of northern Maine, Michigan's Upper Peninsula, the northern lake country of Minnesota, and the Florida Everglades. In the West, the areas of exceptionally high risk include major agricultural areas or areas of combined urban and agricultural cover, such as California's San Joaquin Valley, the Snake River Valley of southern Idaho, the Willamette Valley in Oregon, the Front Range of Colorado, and the Wasatch Front of Utah.

The huge blocks of risk-level 4–5 watersheds in the eastern division reflect, most importantly, the fact that cultivation and high levels of atmospheric

at the 100th meridian. The eastern areas were generally at a higher risk of water-quality impairment than were the deposition (and to a lesser extent, high levels of livestock grazing) cover large expanses of the eastern division,

whereas most other stressors (especially mining, confined animal feeding, and high-density housing) tend to occur in localized areas (figure 2).

Looking at the full set of watersheds, for all three problems, most watersheds were found at the medium risk levels (2–4), with very low-risk and very high-risk watersheds being less common (table 4). For sediments and toxics, fewer than 10% of the watersheds were placed in the highest risk category.

Based on WRR aggregate risk values, the WRRs fall roughly into three groups. The five WRRs at greatest risk, with risk values above .5, are in the Northeast, Midwest, and Tennessee Valley (table 5). The five lowest-risk WRRs, with risk values below .15, are in the Northwest and Intermountain West. The remaining eight WRRs, with risk values from .28 to .45, include New England, the South, California, the southern Plains, and the northern Plains and nearby territory of the Missouri and Souris-Red-Rainy basins.

Table 4. Distribution of watershed risk levels by problem.

Risk level	Sediment	Nutrients	Toxics	All three problems
1	2459	2683	2418	2370
2	3865	3170	3482	3508
3	4069	2989	3727	3542
4	3786	4142	4331	4561
5	1093	2288	1314	1291
All five risk levels	15,272	15,272	15,272	15,272

Table 5. Summary statistics by water resource region.

Water resource region	Risk value	Coefficient of variation of risk value
1. New England	.30	.39
2. Mid-Atlantic	.67	.20
3. South Atlantic Gulf	.45	.21
4. Great Lakes	.53	.31
5. Ohio	.58	.17
6. Tennessee	.49	.17
7. Upper Mississippi	.66	.16
8. Lower Mississippi	.37	.23
9. Souris-Red-Rainy	.32	.30
10. Missouri	.32	.46
11. Arkansas White Red	.43	.32
12. Texas Gulf	.38	.28
13. Rio Grande	.08	.60
14. Upper Colorado	.06	.55
15. Lower Colorado	.10	.67
16. Great Basin	.05	.76
17. Pacific Northwest	.15	.51
18. California	.28	.51

Figure 3 reveals greater heterogeneity in the West than in the East, with pockets of high-risk watersheds in the West often surrounded by vast low-risk areas. This heterogeneity is indicated by the coefficient of variation of the risk values of the watersheds, which exceeds .45 in WRRs 10 and 13–18 but is elsewhere below .4 (table 5). The greater heterogeneity in the West reflects the West's climatic and topographic variability, its heavy reliance on irrigated agriculture (which is restricted to areas receiving pumped or diverted water), and its land-ownership patterns (with wide expanses in public ownership).

As was indicated by the correlations of risk values across problems—.86 between sediment and nutrients, .91 between sediment and toxics, and .93 between nutrients and toxics—high risk levels for one problem tend to be associated with high levels of the other problems. These strong correlations result from at least two factors. First, some stressors are associated with multiple problems. For example, cultivation is a source of sediments, nutrients, and toxics (table 3). Second, different stressors may naturally occur in proximity. For example, housing requires roads, and conditions favoring cultivation also favor livestock grazing. The strong intercorrelations among the three problems lend support for summarizing risk is a single measure, as was done here (figure 3).

To investigate the sensitivity of the overall risk values to the weights, the analysis was rerun using equal weights assigned to all stressors of a given problem. The correlation of the risk values obtained using the PCA-based weights to the risk values that result when equal weights are used is .99. Regression of the equal weight-based risk values on the PCA-based risk values yielded intercept and slope coefficients of −0.02 and 1.06, respectively, which reveals only a slight shift and indicates that the two approaches yield very similar risk values. Therefore, as Falcone and colleagues (2010) found, when several indicators are used, the assignment of weights may be of little significance.

Having specified the causal variables (i.e., stressors) a priori, and lacking an independent measure of risk of impaired water quality, we cannot now test the veracity of our measurements of overall risk. It is interesting nonetheless to note the association of risk with a few measures not included as stressors. Those measures, and the correlations of the measures with risk value, are as follows: the percentage of the watersheds in forest cover (.05), the percentage in range cover (−.70), the percentage in water or wetland cover (.11), the percentage in federal ownership (−.65), mean annual precipitation (.46), and mean elevation (−.68) (see table 1 for the WRR levels of these measures). Therefore, for the United States as a whole, risk is only marginally correlated with forest cover or water or wetland cover but is strongly (negatively) correlated with range cover, land protection, dryness, and elevation. Most of these findings are no surprise, since dryness and rangeland cover go hand in hand in limiting cultivation; federal land ownership restricts cultivation, house construction, animal feeding, and in some instances mining as well; and higher elevations are mostly found in the West, often on public land.

The lack of a strong correlation of percentage forest cover to risk is at first puzzling, because forest cover is generally associated with low risk. Examining WRRs individually revealed that the .05 nationwide correlation results from a balance between regions where the correlation is significantly negative and other regions where it is significantly positive. For most eastern WRRs, the correlation is below −.4, because nonforested areas tend to be agricultural and therefore at greater risk than forested areas. However, in most western WRRs (with the exception of the two West Coast WRRs) the correlation was positive (but always below .4) because nonforested areas tend to be rangeland, which typically has even lower levels of population, roads, and other stressors than do forest areas.

Conclusions

The striking divide at roughly the 100th meridian between the relatively high-risk eastern division and the generally lower-risk western division is related, first and foremost, to precipitation and topography. The wetter climate and more gentle slopes of the East naturally support more cultivation and livestock grazing and, therefore, historically supported greater population densities. The greater population densities in turn led to higher electricity demand and therefore—all else equal—to heavier levels of atmospheric deposition. Conversely, the dryness and dramatic topography of much of the West restricted settlement and, once the conservation movement gained sway, allowed for much more land protection in the West than was feasible in the East. As was wonderfully recounted by Stegner (1954), John Wesley Powell forcefully argued over 130 years ago that much of the West would not support concentrated agriculture and successful settlement. Powell was of course correct about agriculture and would not be surprised by the results presented here.

The snapshot of the relative risk of nonpoint-source water-quality impairment in figure 3 begs the following questions: How has the risk changed in the past, and how is it likely to change in the future? Comparable spatially explicit nationwide data sets of historical levels are not available for some of the stressors used here, so creation of a map comparable to figure 3 for, say, 1950 or 1980 is not possible. However, recent large-scale trends in the levels of stressors offer some insights into how the risk of water-quality problems may have recently changed. They reveal an equivocal picture. Some factors that suggest increasing risk are the steady rise in population (by an average of 2.6 million people per year from 1960 to 2000) and the increase in the area of housing (from 5.9 million to 12.2 million hectares over the same period) (Theobald 2001). Suggesting decreasing risk, total farmed area dropped steadily from a high in 1950 of roughly 469 million hectares to about 376 million hectares in 1997 (NASS 2011), total pesticide use declined during the 1980s and remained flat during the 1990s (Aspelin 2003), total cattle and sheep inventories peaked in the mid-1990s and have since been dropping (although per capita poultry consumption has been rising consistently) (NASS 2011),

and total atmospheric deposition of nitrates and sulfates has dropped significantly over the past two decades in the Northeast and Midwest (USEPA 2010). Finally, the application of nutrients in commercial fertilizers has remained at about 150 kilograms per hectare per year since the mid-1990s, in contrast to the three-fold increase that occurred from 1960 to the mid-1990s (USEPA 2011b). The recent decreasing trends in stressors are encouraging, but it is difficult to know whether the improvements have been sufficient to compensate, on average, for the effects of the inexorable growth in population and related housing and vehicle traffic.

The future trends in stressors are also a complex mixture, as a few examples will show. The Census Bureau expects US population to continue rising at about the past rate over the next 50 years, and the expanding population will require houses (Theobald 2005) and roads (as well as new mines). The expansion of housing and continued improvement in agricultural efficiency will tend to reduce farmed area, although other trends, such as the growth in biofuels, are introducing pressures to bring marginal farm lands into production (Malcolm et al. 2009), leaving the future trend in total agricultural area uncertain. And although changing tastes for red meat (Haley 2001) may contain the growth in livestock grazing, compensating increases in consumption of foul and farm-grown fish will introduce new pressures. Therefore, the net effect on the risk of water-quality impairment of such trends is uncertain, which suggests that continued vigilance in environmental protection may be needed to avoid further deterioration.

An attractive feature of the assessments of watershed condition or of risk of environmental impairment is the possibility of periodic reassessments, which allow the measurement of trends. Indeed, trend analysis is perhaps the most useful role of such an assessment. Although historical data sets do not allow us to replicate the current assessment for past decades, future data sets are likely to enable careful documentation of trends in risk as we move forward.

As was mentioned earlier, cultivated and urbanized areas are prominent sources of nonpoint-source pollution, in contrast to forests, which yield relatively clean water. Therefore, it is instructive to consider how much of our renewable water supplies (estimated as precipitation minus natural evapotranspiration) originate on these cover types. In the western third of the United States (WRRs 13–18), where most precipitation occurs in higher, cooler areas, forests account for 68% of the supplies and rangelands for another 20%, whereas cultivated areas account for only 5% of the water supply (Brown et al. 2008). However, in the middle third of country (WRRs 7–12), where precipitation tends to be distributed more evenly across the landscape, cultivated areas are the predominant source of water, accounting in aggregate for 48% of the water supply, whereas forests are the source of only 27%. And in the eastern third (WRRs 1–6), cultivated lands are the second most prevalent water source (after forests), accounting for 25% of the supply. (Separate estimates for urban and suburban lands were not

available, but they occupy relatively little area.) Therefore, in contrast with the western third of the country, other regions are not only more agricultural, but their cultivated areas are relatively more important as sources of renewable water supply, which raises the importance of addressing nonpoint sources of pollution in those areas.

The confluence of high rates of cultivation, housing, roads, and livestock grazing and feeding in much of the eastern division and in isolated areas of the western division—exacerbated by higher rates of atmospheric deposition in the eastern division—presents a serious resource-management challenge. The recent water-quality summaries and surveys cited above suggest that this challenge has yet to be met for nonpoint-source pollution—by and large in contrast to successes in controlling point-source pollution. Although the Clean Water Act of 1972 addressed both kinds of pollution, it provided mandatory federal regulation only for point-source pollution (Glicksman and Batzel 2010). Making significant progress on controlling nonpoint-source pollution will clearly require renewed energy and focus and perhaps additional Congressional direction as well.

Acknowledgments

References cited

Albasel N, Cottenie A. 1985. Heavy metal contamination near major highways, industrial and urban areas in Belgian grassland. Water, Air, and Soil Pollution 24: 103–109.

Aspelin AL. 2003. Pesticide usage in the United States: Trends during the 20th century. Center for Integrated Pest Management, North Carolina State University. Report no. 105.

Belsky AJ, Matzke A, Uselman S. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54: 419–431.

Benavides-Solorio J, MacDonald LH. 2001. Post-fire runoff and erosion from simulated rainfall on small plots, Colorado Front Range. Hydrological Processes 15: 2391–2412.

Brown TC, Binkley D. 1994. Effect of Management on Water Quality in North American Forests. US Department of Agriculture, Forest Service, Rocky Mountain Research Station. General Technical Report no. RM-248.

Brown TC, Froemke P. 2010. Risk of impaired condition of watersheds containing national forest lands. US Department of Agriculture, Forest Service, Rocky Mountain Research Station. General Technical Report no. RMRS-251.

Brown TC, Hobbins MT, Ramirez JA. 2008. Spatial distribution of water supply in the coterminous United States. Journal of the American Water Resources Association 44: 1474–1487.

Burns DA, McHale MR, Driscoll CT, Roy KM. 2006. Response of surface water chemistry to reduced levels of acid deposition: Comparison of trends in two regions of New York, USA. Hydrological Processes 20: 1611–1627.

Cannon SH, Bigio ER, Mine E. 2001. A process for fire-related debris flow initiation, Cerro Grande Fire, New Mexico. Hydrological Processes 15: 3011–3023.

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8: 559–568.

Clow DW, Mast MA. 1999. Long-term trends in stream water and precipitation chemistry at five headwater basins in the northeastern United States. Water Resources Research 35: 541–554.

Courtney LA, Clements WH. 2002. Assessing the influence of water and substratum quality on benthic macroinvertebrate communities in a metal-polluted stream: An experimental approach. Freshwater Biology 47: 1766–1778.

Danz NP, et al. 2007. Integrated measures of anthropogenic stress in the Great Lakes Basin. Environmental Management 39: 631–647.

Driscoll CT, et al. 2003. Nitrogen pollution in the northeastern United States: Sources, effects, and management options. BioScience 53: 357–374.

Dubrovsky NM, Hamilton PA. 2010. The quality of our nation's waters—Nutrients in the nation's streams and groundwater: National findings and implications. US Geological Survey. Fact sheet 2010-3078.

Falcone JA, Carlisle DM, Weber LC. 2010. Quantifying human disturbance in watersheds: Variable selection and performance of a GIS-based disturbance index for predicting the biological condition of perennial streams. Ecological Indicators 10: 264–273.

Forman RTT, Alexander LE. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29: 207–231.

Gianessi LP, Peskin HM, Puffer CA. 1986. National data base of nonurban-nonpoint-source discharges and their effect on the nation's water quality. Resources for the Future. Technical Report no. PB-87-137972/XAB.

Glicksman RL, Batzel MR. 2010. Science, politics, law and the arc of the Clean Water Act: The role of assumptions in the adoption of a pollution control landmark. Washington University Journal of Law and Policy 32: 99–138.

Haley MM. 2001. Changing consumer demand for meat: The U.S. example, 1970–2000. Pages 41–48 in Regni A, ed. Changing Structure of Global Food Consumption and Trade. US Department of Agriculture, Economic Research Service. Agriculture and Trade Report no. WRS-01-1.

Hoffman RS, Capel PD, Larson SJ. 2000. Comparison of pesticides in eight U.S. urban streams. Environmental Toxicology and Chemistry 19: 2249–2258.

Jones KB, Ritters KH, Wichham JD, Tankersley RD Jr, O'Neill RB, Chaloud DJ, Smith ER, Neale AC. 1997. An ecological assessment of the United States Mid-Atlantic region: A landscape atlas. US Environmental Protection Agency. Report no. EPA/600/R-97/130.

Jones KB, Neale AC, Nash MS, van Remortel RD, Wickham JD, Riitters KH, O'Neill RV. 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: A multiple watershed study from the United States Mid-Atlantic region. Landscape Ecology 16: 301–312.

Kauffman JB, Krueger WC. 1984. Livestock impacts on riparian ecosystems and streamside management implications... A review. Journal of Range Management 37: 430–438.

Lettenmaier DP, Hooper ER, Wagoner C, Faris KB. 1991. Trends in stream quality in the continental United States, 1978–1987. Water Resources Research 27: 327–339.

Malcolm SA, Aillery M, Weinberg M. 2009. Ethanol and a Changing Agricultural Landscape. US Department of Agriculture, Economic Research Service. Economic Research Report no. 86.

Mallin MA. 2000. Impacts of industrial animal production on rivers and estuaries. American Scientist 88: 26–37.

Moody JA, Martin DA. 2001. Initial hydrologic and geomorphic response following a wildfire in the Colorado Front Range. Earth Surface Processes and Landforms 26: 1049–1070.

Mueller DK, Helsel DR. 1996. Nutrients in the Nation's Waters: Too Much of a Good Thing? US Geological Survey. Circular no. 1136.

[NASS] National Agricultural Statistics Service. 2011. Trends in U.S. agriculture. NASS. (20 November 2011; *www.nass.usda.gov/Publications/Trends_in_U.S._Agriculture/index.asp*)

National Research Council. 2010. Toward sustainable agricultural systems in the 21st century. National Academies Press.

Neary DG, Ryan KC, DeBano LF, eds. 2005. Wildland Fire in Ecosystems: Effects of Fire on Soil and Water. US Department of Agriculture, Forest

Service, Rocky Mountain Research Station. General Technical Report no. RMRS-42, vol. 4.

[NRCS] US Department of Agriculture Natural Resources Conservation Service. 2009. Geospatial Data Gateway. NRCS. (20 November 2011; *http://datagateway.nrcs.usda.gov/GDGHome_StatusMaps.aspx*)

———. 2011. Information about hydrologic units and the watershed boundary dataset. National Geospatial Management Center. (20 November 2011; *www.nrcs.usda.gov/wps/portal/nrcs/detail/national/ngmc/?&cid=nrcs143_021616*)

Omernik JM. 1977. Nonpoint Source-stream Nutrient Level Relationships: A Nationwide Study. US Environmental Protection Agency, Environmental Research Laboratory, Office of Research and Development. Report no. 600/3-77-105.

Paul MJ, Meyer JL. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32: 333–365.

Paulson SG, et al. 2008. Condition of stream ecosystems in the US: An overview of the first national assessment. Journal of the North American Benthological Society 27: 812–821.

Primpas I, Tsirtsis G, Karydis M, Kokkoris GD. 2010. Principal component analysis: Development of a multivariate index for assessing eutrophication according to the European water framework directive. Ecological Indicators 10: 178–183.

Roline RA. 1988. The effects of heavy metals pollution of the upper Arkansas River on the distribution of aquatic macroinvertebrates. Hydrobiology 160: 3–8.

Rösner U. 1998. Effects of historical mining activities on surface water and groundwater—an example from northwest Arizona. Environmental Geology 33: 224–230.

Seilheimer TS, Mahoney TP, Chow-Fraser P. 2009. Comparative study of ecological indices assesssing human-induced disturbance in coastal wetlands of the Laurentian Great Lakes. Ecological Indicators 9: 81–91.

Smith RA, Alexander RB, Wolman MG. 1987. Water-quality trends in the nation's rivers. Science 235: 1607–1615.

Stein JL, Stein JA, Nix HA. 2002. Spatial analysis of anthropogenic river disturbance at regional and continental scales: Identifying the wild rivers of Australia. Landscape and Urban Planning 60: 1–25.

Stegner W. 1954. Beyond the Hundredth Meridian: John Wesley Powell and the Second Opening of the West. Penguin.

Stoddard JL, et al. 1999. Regional trends in aquatic recovery from acidification in North America and Europe. Nature 401: 575–578.

Swetnam TW, Allen CD, Betancourt JL. 1999. Applied historical ecology: Using the past to manage the future. Ecological Applications 9: 1189–1206.

Theobald DM. 2001. Land-use dynamics beyond the American urban fringe. Geographical Review 91: 544–564.

———. 2005. Landscape patterns of exurban growth in the USA from 1980 to 2020. Ecology and Society 10 (Art. 32). (20 November 2011; *www.ecologyandsociety.org/vol10/iss1/art32/ES-2005-1390.pdf*).

[USEPA] Environmental Protection Agency. 2009. National Water Quality Inventory: Report to Congress, 2004 Reporting Cycle. USEPA, Office of Water. Report no. EPA-841-R-08-001.

———. 2010. Our nation's air—Status and trends through 2008. USEPA. (20 November 2011; *www.epa.gov/airtrends/2010*)

———. 2011a. Water quality assessment and total maximum daily loads information. USEPA. (20 November 2011; *www.epa.gov/waters/ir*)

———. 2011b. Report on the environment. USEPA. (20 November 2011; *www.epa.gov/ncea/roe*)

[USGS] US Geological Survey. 2009. Hydrologic unit map. (20 November 2011; *http://water.usgs.gov/GIS/regions.html*)

Vörösmarty CJ, et al. 2010. Global threats to human water security and river biodiversity. Nature 467: 555–561.

Wang L, Brenden T, Seelbach P, Cooper A, Allan D, Clark R Jr, Wiley M. 2008. Landscape based identification of human disturbance gradients and reference conditions for Michigan streams. Environmental Monitoring and Assessment 141: 1–17.

Wohl NE, Carline RF. 1996. Relations among riparian grazing, sediment loads, macroinvertebrates, and fishes in three central Pennsylvania streams. Canadian Journal of Fisheries and Aquatic Sciences 53: 260–266.

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