EXHIBIT 1

0 2.5 5 10 15 20 Miles

Anadromous Streams
AWC Stat Area
Regional Boundary
1:63360 Quadrangle Boundary

Labeled Quadrangles indicate quadrangles that contain AWC information.

Waters important to Anadromous Fish are listed pursuant to AS 16.05.871. Specified species distribution and life functions reflect known data. Actual distribution and use may extend beyond specified limits. Migration upstream and/or downstream is assumed for specified stream reaches.

Map Location

Produced By
State of Alaska
Department of Fish and Game

Anadromous Waters Atlas

Quad No 059

Tyonek
Index

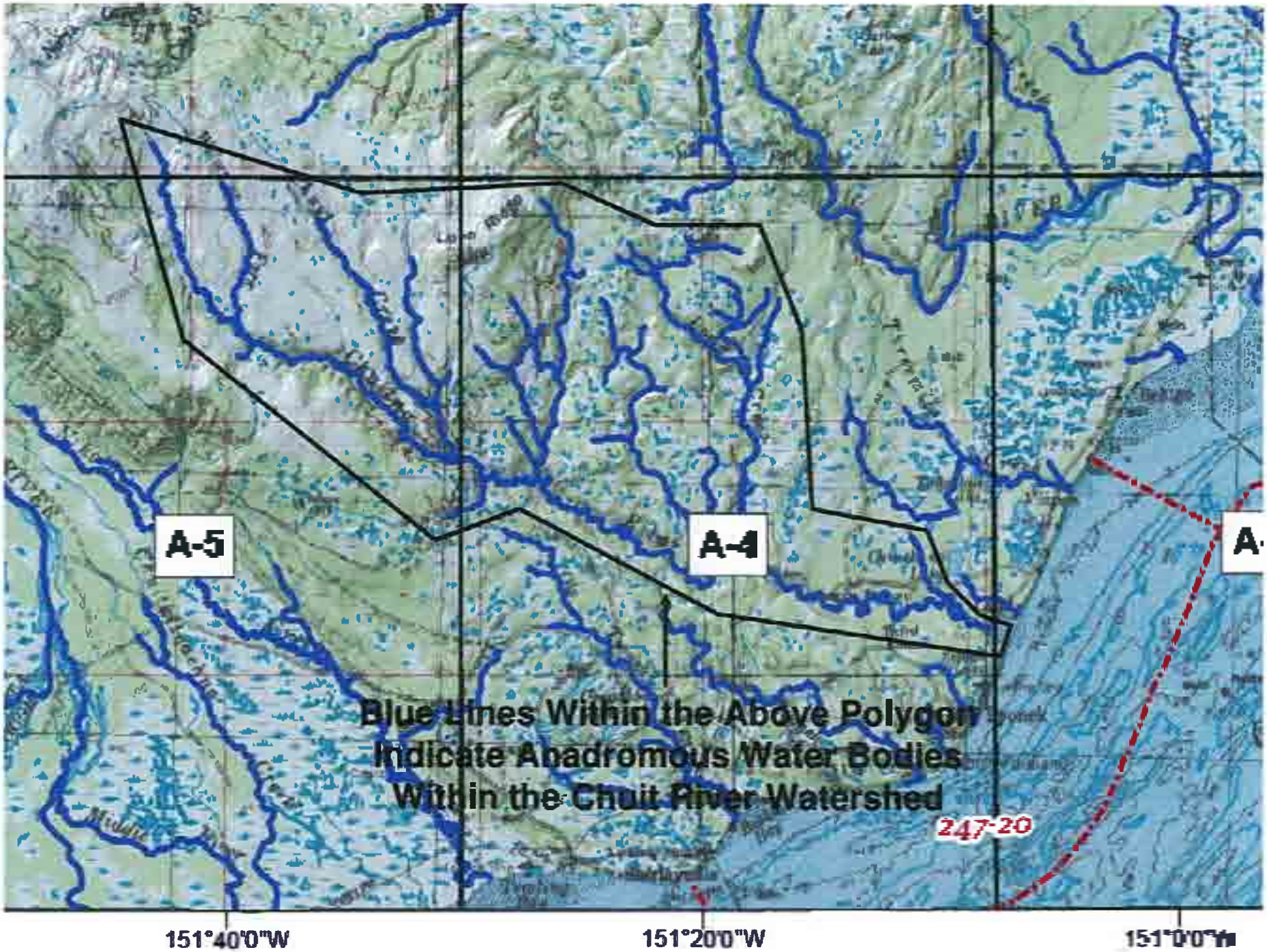
A-5
A-4
A-
Blue Lines Within the Above Polygon
Indicate Anadromous Water Bodies
Within the Chuit River Watershed
247-20
151°40'0"W
151°20'0"W
151°0'0"W

EXHIBIT 2

STATE OF ALASKA
DEPARTMENT OF NATURAL RESOURCES
DIVISION OF LAND AND WATER MANAGEMENT

OFFICE USE ONLY

[illegible]

Southcentral District
Pouch 7-005
Anchorage, Alaska 99510
(907) 276-2653

Southeastern District
230 S. Franklin, Room 407
Juneau, Alaska 99801
(907) 465-3400

LAS

APPLICATION FOR RESERVATION OF WATER

INSTRUCTIONS: This is an application to reserve a specific instream flow or level of water under AS 46.15.145 and 11 AAC 93.141-147. This application must be filled out completely and all requested attachments submitted with it. Failure to complete all parts of the application may result in return of the application. Attach extra pages to fully answer questions. If a report is attached as part of this application, indicate appropriate page numbers following each question. Submit this application to the district in which the proposed reservation is located (identified above). Please type or print in ink.

1. Full legal name of applicant(s): ALASKA DEPARTMENT OF FISH AND GAME

2. Mailing Address: P.O. BOX 25526

City: JUNEAU State: AK Zip: 99802-5526

Business Phone: 907-465-4100 Home Phone: _____

3. Name of the stream or water body in which water is proposed to be reserved:

CHUITNA RIVER

4. Location of the proposed reservation of water:

(a) List ALL sections, townships, ranges and meridians from the beginning to the end of the stream segment and for all parts of the lake or waterbody in which water is requested to be reserved. (Attach extra pages if needed.)

SEE ATTACHED APPENDIX A

1151 (11/83)

(b) Describe the location of the point or points defining the boundary of the proposed reservation of water by river mile index, river mile, geographical or cultural landmark, etc., on the stream of water body. (Attach extra pages if needed.)

SEE ATTACHED APPENDIX A

(c) ATTACH a U.S. Geological Survey map at 1:63,360 scale, or 1:250,000 scale if 1:63,360 scale is unavailable for the area, clearly identifying the following for the proposed reservation of water:

SEE ATTACHED APPENDIX A

(1) Sections, townships, ranges and meridians

(2) The stream or water body in which the reservation of water is proposed

(3) Specific point or points defining the boundary of the proposed reservation of water

(4) Permanent, temporary or planned locations of water measurement devices (such as gaging stations, weirs, staff gages)

(5) Permanent, temporary or planned bench marks

5. (a) Identify the purpose(s) of the proposed reservation of water by checking the appropriate box(es).

[X] protection of fish and wildlife habitat, migration, and propagation

[ ] recreation and park purposes

[ ] navigation and transportation purposes

[ ] sanitary and water quality purposes

(b) Describe in detail the purpose(s) of the proposed reservation, including, when appropriate; species and life stage, type of recreation, vehicle, or water quality parameter, or other relevant information. (Attach extra page if needed.)

SEE ATTACHED APPENDIX A

6. Is the water currently being used for the purpose(s) applied for?

[X] Yes.

[ ] No. If no, when will use for this purpose begin? Specify approximate date.

_____

7. (a) Water requested to be reserved (check one):

[X] to maintain a specific instream flow rate, measured in cubic feet per second

[ ] to maintain a specific level of surface water, measured in cubic feet or acre feet

[ ] to maintain a specific surface water elevation, measured in relation to a permanent benchmark.

(b) Quantify the specific amount of water requested to be reserved: Identify and quantify, as appropriate, flow rates, quantities, surface water elevations, depths, etc., as they relate to the daily durations and months of the year during which the reservation is proposed. Include any flow release schedules from projects upstream of the proposed reservation that would apply. (Attach extra pages if needed.)

SEE ATTACHED APPENDIX A

8. Attach and submit with this application documentation or reports showing facts to support the following:

(a) The need for the proposed reservation of water, including reasons why this reservation is being requested. SEE ATTACHED APPENDIX A

(b) Identify and describe the methodology, data, and data analysis used to substantiate the need for and the quantity of water requested for the proposed reservation of water, including:

SEE ATTACHED APPENDICES A, B, C, D.

(1) Name and description of method used,

(2) Who conducted the study and analysis,

(3) Schedule of when data collection and analysis occurred,

(4) Type(s) of instrument(s) used to collect and analyze data,

(5) Description of data and how the data was collected, including when applicable, (A) selection of stream reach, study site and transect selection, (B) flow, survey, elevation, and depth measurements, (C) pertinent physical, biological, water chemistry and socio-economic data,

(6) Description of how data was analyzed, and

(7) Maps, photos, aerial photos, calculations, and any other documents supporting this application.

9. If there are provisions for monitoring this proposed reservation of water, include the following:

SEE ATTACHED APPENDIX A

(a) Description of monitoring equipment (such as gaging stations, staff gages, weirs):

(b) Location of monitoring equipment

(c) Provisions for payment of monitoring

(d) Reporting system.

Statements contained in this application are true and correct to the best of my knowledge.

Signed _____

LARRY L. TRAUT, FOR THE ALASKA DEPARTMENT OF FISH & GAME

Applicant(s) Full Legal Name(s)

Date 5/16/90

Address all questions pertaining to this application to Christopher Estes, Statewide Instream Flow Coordinator, Alaska Department of Fish and Game, 267-2142.

1151 (11/83)

EXHIBIT 3

Annu. Rev. Ecol. Evol. Syst. 2004. 35:257–84
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LANDSCAPES AND RIVERSCAPES: The Influence of Land Use on Stream Ecosystems

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Key Words catchment, disturbance, stressor response, stream health, river

■ **Abstract** Local habitat and biological diversity of streams and rivers are strongly influenced by landform and land use within the surrounding valley at multiple scales. However, empirical associations between land use and stream response only varyingly succeed in implicating pathways of influence. This is the case for a number of reasons, including (*a*) covariation of anthropogenic and natural gradients in the landscape; (*b*) the existence of multiple, scale-dependent mechanisms; (*c*) nonlinear responses; and (*d*) the difficulties of separating present-day from historical influences. Further research is needed that examines responses to land use under different management strategies and that employs response variables that have greater diagnostic value than many of the aggregated measures in current use.

In every respect, the valley rules the stream.
H.B.N. Hynes (1975)

INTRODUCTION

Rivers are increasingly investigated from a landscape perspective, both as landscapes in their own right (Robinson et al. 2002, Ward 1998, Wiens 1989) and as ecosystems that are strongly influenced by their surroundings at multiple scales (Allan et al. 1997, Fausch et al. 2002, Schlosser 1991, Townsend et al. 2003). River ecologists have long recognized that rivers and streams are influenced by the landscapes through which they flow (Hynes 1975, Vannote et al. 1980). However, a landscape perspective of rivers continues to evolve, owing both to the emergence of landscape ecology as a field of study (Turner et al. 2001, Wiens 1989) and to an increased focus on catchment-scale studies by freshwater ecologists.

As Wiens (2002) observed, river ecologists have been doing landscape ecology for a long time but just not calling it that. Landscape ecology places particular emphasis on habitat heterogeneity, connectivity, and scale, all of which have received considerable attention in running waters (Allan 1995). However, most earlier work was conducted at small spatial scales, often within stream reaches of a few hundred

meters and their immediate surroundings; less consideration was given to the importance of larger spatial units. Our current understanding of rivers, as with other ecosystems, increasingly incorporates a conceptual framework of spatially nested controlling factors in which climate, geology, and topography at large scales influence the geomorphic processes that shape channels at intermediate scales and thereby create and maintain habitat important to the biota at smaller scales (Allen & Starr 1982, Frissell et al. 1986, Snelder & Biggs 2002). Recognizing that rivers are complex mosaics of habitat types and environmental gradients, characterized by high connectivity and spatial complexity, riverine landscapes increasingly are viewed as "riverscapes" (Fausch et al. 2002, Schlosser 1991, Ward et al. 2002), a unit that is amenable to study over a wide range of scales from a braided river and its valley (Tockner et al. 2002) to small habitat patches (Palmer et al. 2000).

Investigators increasingly recognize that human actions at the landscape scale are a principal threat to the ecological integrity of river ecosystems, impacting habitat, water quality, and the biota via numerous and complex pathways (Allan et al. 1997, Strayer et al. 2003, Townsend et al. 2003). In addition to its direct influences, land use interacts with other anthropogenic drivers that affect the health of stream ecosystems, including climate change (Meyer et al. 1999), invasive species (Scott & Helfman 2001), and dams (Nilsson & Berggren 2000). The recent increase in studies that seek to establish relationships between land use and stream condition is driven by several developments: (*a*) the widespread recognition of the extent and significance of changes in land use and land cover worldwide (Meyer & Turner 1994), (*b*) conceptual and methodological advances in landscape ecology combined with the ready availability of land use/land cover data (Turner et al. 2001), and (*c*) the increasing use of indicators of stream health to assess status and trends of rivers (Karr & Chu 2000, Norris & Thoms 1999).

Hierarchies, Habitats, and Biodiversity

An extensive literature explores the hierarchical nature of river systems, from the largest spatial scale of landscape or basin to successively smaller scales of the valley segment, channel reach, individual channel units (such as riffles and pools), and microhabitat (Figure 1) (Fausch et al. 2002, Frissell et al. 1986, Montgomery 1999). Because stream ecosystems are typically characterized by habitat and biota observed at the scale of a reach, typically 10^2 – 10^3 m in length, and local species assemblages are strongly influenced by habitat quality and complexity, this geomorphological framework suggests how the stream environment at the local scale is influenced by the surrounding landscape. Reach-level channel morphology is influenced by valley slope and confinement, bed and bank material, and riparian vegetation, as well as by the supply from upslope of water, sediments, and wood (Montgomery & MacDonald 2002). Many features of the dynamic river channel are mutually adjusting (Church 2002), and human activities on the landscape that affect water or sediment supply or that stabilize or destabilize the existing channel shape are likely to set off a complex cascade of changes that are ultimately manifest

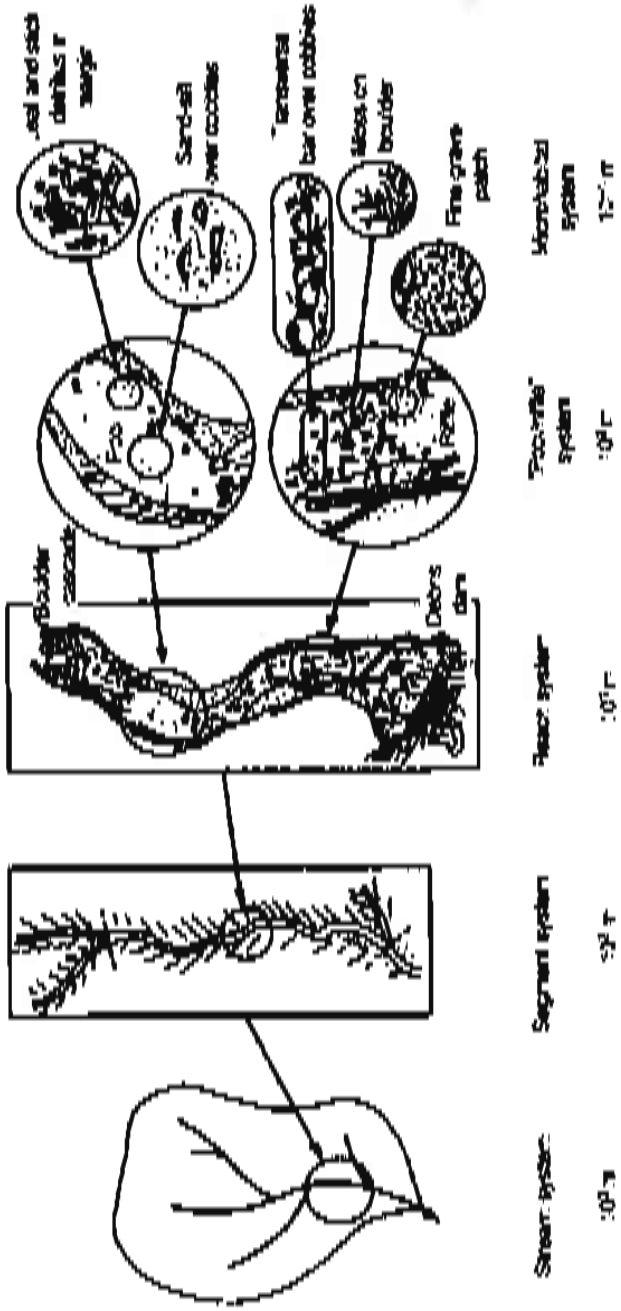

Figure 1 The hierarchical nature of river systems, from the largest spatial scale of landscape or basin to successively smaller scales of the valley segment, channel reach, individual channel units (such as riffles and pools), and microhabitats, as visualized for a small mountain stream. (Frissell et al. 1986) Reprinted with permission from Environmental Management.

in altered and possibly degraded stream habitat. Natural hydrologic variability, and high flows in particular, move and sort sediments, and through cycles of erosion and deposition create a variety of channel features, including riffles, pools, bars, and islands; cause channel migration; maintain floodplain connectivity and other complex elements of floodplain river channels, including meander loops and side channels; and make the land-water interface both complex and dynamic (Junk et al. 1989). The resultant ever-changing mosaic of habitat patches, ecotones, and successional stages—the riverscape in all its complexity—is largely responsible for the high biodiversity of these systems (Robinson et al. 2002, Ward 1998). In this patch dynamics perspective (Townsend 1989, Hildrew & Giller 1994), the interaction between species-specific habitat needs, life histories, and dispersal ability and the ever-shifting temporal and spatial mosaic of stream habitats support greater diversity than would occur were the habitat unchanging. Thus, both the variety and the variability of habitat are important in influencing the biological diversity of streams and are linked to the larger stream system and surrounding landscape. Human actions at the landscape scale disrupt the geomorphic processes that maintain the riverscape and its associated biota and frequently result in habitat that is both degraded and less heterogeneous.

Assessment of River Health

Ecological integrity, stream condition, and river health are terms that describe the status of fluvial ecosystems and their response to human influences. Condition is defined by similarity of a test site to a set of least impaired reference sites, whether measured by the sum of several indicators, such as the number of intolerant species and taxa richness [Index of Biotic Integrity (IBI), see Karr 1991], or by the number of observed taxa relative to expected (Rivpacs, see Wright 1995; Australs, see Norris & Hawkins 2000). Additional measures include taxa richness of sensitive species; various biological and ecological traits, such as body size and shape, life history, and behavioral traits (Townsend & Hildrew 1994, Corkum 1999, Usseglio-Polatera et al. 2000b, Richards et al. 1997, Pan et al. 1999); pollution tolerance (Hilsenhoff 1988); and ecosystem processes, such as photosynthesis and respiration (Bunn et al. 1999). Habitat and water quality are also evaluated using individual variables or combined metrics (Barbour et al. 1999). Thus, a wide variety of assessment methods are available to evaluate the response of stream condition to a gradient in land use.

The shape of the relationship between a stream response variable and a measure of stress (Figure 2) likely depends mutually on the sensitivity of the response variable and mode of action of the environmental stressor. A gradual decline indicating incremental change in stream condition might be expected, for example, if a steady increase in sedimentation acted on a species assemblage that was approximately linearly ranked in their sensitivity to sediments. Nonlinear responses are expected whenever the species in question, or the majority of species, exhibit a sensitivity threshold to a particular stress, such as the frequency or magnitude of high flows. Although the response of the biota to some stressors, such as insecticides,

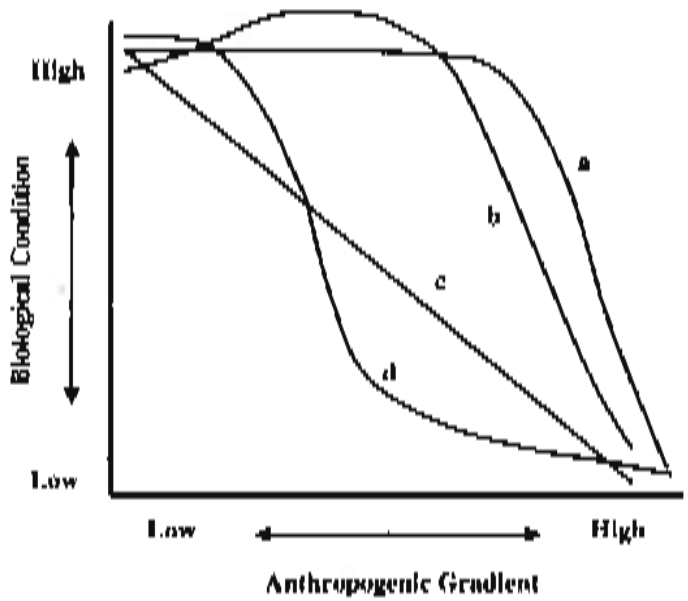

Figure 2 Hypothetical relationship depicting possible responses of stream biological condition (taxon richness, assemblage similarity, or a biological index, scaled to best attainable or reference conditions) to a gradient of increasing environmental stress, measured directly as, e.g., sedimentation, or indirectly as, e.g., agricultural land in the catchment. Possible responses include (*a*) nonlinear response occurring in the high range of the gradient, (*b*) subsidy-stress response, (*c*) linear response, and (*d*) nonlinear (threshold) response occurring in the low range of the gradient. Curves (*a*) versus (*d*) indicate low versus high sensitivity to a stressor. Modified from Norris & Thoms (1999) and Quinn (2000).

is expected to be only negative, a number of environmental stressors have positive influence at low to moderate concentrations. For example, a subsidy stress response (Odum et al. 1979) may be a common outcome of riparian thinning and a low intensity of agriculture, in which initial increases in light, nutrients, and water temperatures increase periphyton biomass and macroinvertebrate abundance, with no apparent decline in diversity, whereas further intensification of agriculture results in loss of diversity and sensitive species (Quinn 2000).

Ultimately, the range of stream conditions from pristine to profoundly impacted reflects the system's integrated response to various human disturbances acting through the physical space of the catchment hierarchy, over short (pulse) and long (press) durations, and with cascading influences via local habitat structure and food web interactions (Quinn 2000, Townsend & Riley 1999). Different disturbances will exert their influence at different spatial scales and by different pathways (Table 1). Because streams are usually affected by multiple and interacting disturbances, matching a response to the responsible stressor can be very

TABLE 1 Principal mechanisms by which land use influences stream ecosystems

Environmental factor	Effects	References
Sedimentation	Increases turbidity, scouring and abrasion; impairs substrate suitability for periphyton and biofilm production, decreases primary production and food quality causing bottom-up effects through food webs; in-filling of interstitial habitat harms crevice occupying invertebrates and gravel-spawning fishes; coats gills and respiratory surfaces; reduces stream depth heterogeneity, leading to decrease in pool species	Burkhead & Jelks 2001, Hancock 2002, Henley et al. 2000, Quinn 2000, Sutherland et al. 2002, Waicer & Hart 1999, Wood & Armitage 1997
Nutrient enrichment	Increases autotrophic biomass and production, resulting in changes to assemblage composition, including proliferation of filamentous algae, particularly if light also increases; accelerates litter breakdown rates and may cause decrease in dissolved oxygen and shift from sensitive species to more tolerant, often non-native species	Carpenter et al. 1998, Deloy & Brusven 1998, Lenat & Crawford 1994, Mainstone & Parr 2002, Niyogi et al. 2003
Contaminant pollution	Increases heavy metals, synthetics, and toxic organics in suspension associated with sediments and in tissues; increases deformities; increases mortality rates and impacts to abundance, drift, and emergence in invertebrates, depresses growth, reproduction, condition, and survival among fishes; disrupts endocrine system; physical avoidance	Clements et al. 2000, Cooper 1993, Kolpin et al. 2002, Liess & Schulz 1999, Rolland 2000, Schulz & Liess 1999, Woodward et al. 1997
Hydrologic alteration	Alters runoff-evapotranspiration balance, causing increases in flood magnitude and frequency, and often lowers base flow; contributes to altered channel dynamics, including increased erosion from channel and surroundings and less-frequent overbank flooding, runoff more efficiently transports nutrients, sediments, and contaminants, thus further degrading in-stream habitat. Strong effects from impervious surfaces and stormwater conveyance in urban catchments and from drainage systems and soil compaction in agricultural catchments	Allan et al. 1997, Paul & Meyer 2001, Poff & Allan 1995, Walsh et al. 2001, Wang et al. 2001
Riparian clearing/canopy opening	Reduces shading, causing increases in stream temperatures, light penetration, and plant growth; decreases bank stability, inputs of litter and wood, and retention of nutrients and contaminants, reduces sediment trapping and increases bank and channel erosion; alters quantity and character of dissolved organic carbon reaching streams; lowers retention of benthic organic matter owing to loss of direct input and retention structures; alters trophic structure	Bourque & Pomeroy 2001, Findlay et al. 2001, Gregory et al. 1991, Gurnell et al. 1995, Lowrance et al. 1984, Martin et al. 1999, Osborne & Kovacic 1993, Stauffer et al. 2000
Loss of large woody debris	Reduces substrate for feeding, attachment, and cover; causes loss of sediment and organic material storage; reduces energy dissipation; alters flow hydraulics and therefore distribution of habitats; reduces bank stability; influences invertebrate and fish diversity and community function	Ehrman & Lamberti 1992, Gurnell et al. 1995, Johnson et al. 2003, Maridet et al. 1995, Stauffer et al. 2000

difficult. Thus, it may be possible to determine the degree of impairment accurately without achieving the same level of certainty regarding cause (Gergel et al. 2002).

THE INFLUENCE OF LAND USE ON RIVERS

The global transition from undisturbed to human-dominated landscapes has impacted ecosystems worldwide and made the quantification of land use/land cover (hereafter, land use) a valuable indicator of the state of ecosystems (Meyer & Turner 1994). Hundreds of studies document statistical associations between land use and measures of stream condition using multisite comparisons and empirical models, and collectively these studies provide strong evidence of the importance of surrounding landscape and human activities to a stream's ecological integrity. Moreover, the extent of land use transformation is staggering. For example, before the development of pastoral agriculture in New Zealand, more than 80% of the land was forested; today, agriculture, primarily the grazing of nearly 60 million sheep and cattle, is the dominant land use in the middle and lower catchment areas of most of New Zealand's streams and rivers (Quinn 2000).

Not surprisingly, agriculture occupies the largest fraction of land area in many developed catchments, whereas urban land use is a much smaller fraction. Of some 150 major river basins of North America, agricultural land use varied from near zero in some northern river systems to 66% in the Upper Mississippi Basin (Benke & Cushing 2004). Six major river basins of the United States have more than 40% of their area in agriculture: the Lower Mississippi, Upper Mississippi, Southern Plains, Ohio, Missouri, and Colorado. Within the Upper Mississippi, the extent of agriculture in large tributary basins varies from 25% in the St. Croix and Wisconsin Rivers to 95% in the Minnesota River Basin. Comparisons of small subcatchments within a larger catchment have reported that the extent of agricultural land use varies even more widely at this smaller spatial scale, from 10% to 70% (Roy et al. 2003), 14% to 99%, (Richards et al. 1996), and 36% to 84% (Roth et al. 1996). Streams draining these landscapes can be expected to experience a wide range of human influences.

Urban land use is commonly a low percentage of total catchment area, yet it exerts a disproportionately large influence both proximately and over distance (Paul & Meyer 2001). Urban land exceeds 5% of catchment area in 29 river basins and exceeds 10% in only 10 of the 150 large basins of North America (Benke & Cushing 2004). However, a large percentage of the land area of small catchments may be urban. Among 30 small (100 km^2) subcatchments of the Etowah Basin, Georgia, combined low- and high-density urban land area averages 15%, with a maximum of 61% (Roy et al. 2003). Impervious surface area reaches as high as 51% for small streams in metropolitan subcatchments of Melbourne, Australia (Walsh et al. 2001), and urban land area is as high as 97% in small catchments of southeastern Wisconsin (Wang et al. 2001).

Other land uses affect stream condition, including forestry, mining, and recreation (Bryce et al. 1999). However, most landscape-scale studies of the influence

of land use on streams have contrasted the varying extent of agricultural, urban, and natural (usually forested) land, and so these studies are the primary focus of this review.

Some important caveats apply to studies of the relationship between land use and stream condition. Because land use sums to 100%, several measures of land use may predict stream condition nearly equally well (e.g., Herlihy et al. 1998), and so the interpretation that a particular land use variable is the primary driver of stream condition must be made with caution. Comparisons of land use implicitly substitute space for time, as the often unstated assumption is that locations differing in land use are similar in essentially all other respects and can be viewed as equivalent to the progression over time of a single location experiencing the transition from natural to developed land. Forecasting changes in stream ecosystems in response to changing land use runs the risk that the relationship will change over time owing to changes in specific practices or in the environment itself. For example, revegetation of the riparian to reduce stream temperatures may be negated by future climate change; development of crops with engineered pest resistance may reduce use of pesticides, thus removing one of the pathways by which agricultural land use impacts stream biota.

Agricultural Land Use

Numerous studies have documented declines in water quality, habitat, and biological assemblages as the extent of agricultural land increases within catchments (Richards et al. 1996, Roth et al. 1996, Sponseller et al. 2001, Wang et al. 1997). Researchers commonly report that streams draining agricultural lands support fewer species of sensitive insect and fish taxa than streams draining forested catchments (Cenito et al. 2002, Lenat & Crawford 1994, Wang et al. 1997). Although researchers report that row crop and other forms of intensive cultivation strongly affect stream condition, the influence of pasture agriculture may be less pronounced (Meador & Goldstein 2003, Strayer et al. 2003).

Agricultural land use degrades streams by increasing nonpoint inputs of pollutants, impacting riparian and stream channel habitat, and altering flows (Table 1). Higher inputs of sediments, nutrients, and pesticides accompany increased agricultural land use (Cooper 1993, Johnson et al. 1997, Lenat 1984, Osborne & Wiley 1988). Landscape metrics, particularly the proportion of agriculture in the catchment and forest in the riparian zone, explained 65%–84% of the variation in yields of nitrogen, dissolved phosphorus, and suspended sediments for 78 catchments across the five-state Mid-Atlantic Highlands region (Jones et al. 2001). Elevated nutrient concentrations are reported to result in greater algal production and changes in autotroph assemblage composition (Delong & Brusven 1998, Quinn 2000). However, the hypoxic conditions that high nutrient loading causes in lentic and coastal waters (Carpenter et al. 1998) are uncommon in streams and are likely to occur only in localized areas of slow-moving water. Because light levels, nutrient concentrations, and water temperature all tend to

increase as riparian forest is lost, algal response may be influenced by one or more of these factors acting in concert. Changes in algal biomass and composition in the upper Roanoke Basin were primarily attributed to light and temperature because nutrients were thought to be sufficient at all sites (Sponseller et al. 2001). Another common response to lost riparian forest is increased macroinvertebrate abundance, particularly grazers, as the food web becomes increasing influenced by autochthonous rather than allochthonous energy sources (Delong & Brusven 1998, Quinn 2000).

Agricultural insecticide and herbicide runoff is likely responsible for some of the association between agricultural land use and stream biota described above (Cooper 1993, Skinner et al. 1997); however, evidence comes primarily from localized toxicity tests rather than from landscape-scale investigations. For example, field enclosures using caged amphipods and laboratory tests that exposed midge larvae to stream sediments showed pesticide toxicity in an agricultural catchment in the United Kingdom (Crane et al. 1996). Furthermore, the disappearance of 8 of the 11 most abundant invertebrate taxa from a reach of headwater stream after surface runoff from arable land was attributed to an insecticide (Schulz & Liess 1999), although most species recovered within 6–11 months, indicating a pulse disturbance. Because the concentrations of agricultural pesticides and herbicides are seldom measured in studies relating agricultural land use to stream biota, their role may be more widespread than is recognized.

Streams in highly agricultural landscapes tend to have poor habitat quality, reflected in declines in habitat indexes and bank stability (Richards et al. 1996, Roth et al. 1996, Wang et al. 1997), as well as greater deposition of sediments on and within the streambed. Sediments in runoff from cultivated land and livestock trampling (Quinn 2000, Strand & Merritt 1999) are considered to be particularly influential in stream impairment (Waters 1995). In the Piedmont region of the Chattahoochee Basin, Georgia, sediments in the channel increased with increasing agricultural land use, while heterogeneity in stream depth and the diversity of fishes associated with coarse substrate in pools declined (Walser & Bart 1999).

Changes to stream hydrology owing to increased agricultural land use are variable, depending on crop evapotranspiration rates compared with natural vegetation, changes to soil infiltration capacity, extent of drainage systems, and, if there is irrigation, whether water is extracted from the river or from groundwater. Mean annual flow of the Kankakee River, Illinois, increased during the twentieth century without any corresponding trend in precipitation, implicating land clearing and urbanization as the cause of greater runoff (Peterson & Kwak 1999). Storm flows commonly increase in magnitude and frequency, especially where runoff is enhanced owing to drainage ditches, subsurface drains, and loss of wetland area. In addition to the impact of flow extremes on erosion and habitat, high flows can eliminate taxa if such events occur during sensitive life stages or with sufficient frequency that only resistant and rapidly dispersing species can tolerate them. Macroinvertebrates that are able to withstand dislodgement or that have short and fast life cycles and

good colonizing ability predominated in highly agricultural streams of Michigan (Richards et al. 1997). Alterations to flow regime affect stream fishes by downstream displacement of early life stages and disruption of spawning (Harvey 1987, Schlosser 1985). Although annual and storm flows typically increase with agricultural land use, base flows often decline owing to reduced infiltration and more episodic export of water (Poff et al. 1997). This decline results in an increased area of shallow water habitat, which usually lacks structure and is more easily warmed (Richards et al. 1996).

Wherever agriculture or other anthropogenic activity extends to the stream margin and natural riparian forest is removed, streams are usually warmer during summer and receive fewer energy inputs as leaf litter, and primary production usually increases (Quinn 2000). Bank stability may decrease, although establishment of deep rooting grasses can stabilize banks (Davies-Colley 1997, Lyons et al. 2000), and the amount of large wood in the stream declines markedly (Johnson et al. 2003). Stable wood substrate in streams performs multiple functions, influencing channel features and local flow and habitat and providing cover for fish, perching habitat for invertebrates, and a substrate for biofilm and algal colonization (Gregory et al. 2003). Its absence can have a profound influence. For example, the presence of wood added an average of 55% and 26% to reach-level local diversity within highly agricultural catchments in Minnesota and Michigan, respectively (Johnson et al. 2003).

Urban Land Use

Substantial changes in biological assemblages are associated with increasing catchment area as urban land (Booth & Jackson 1997, Klauda et al. 1998, Lenat & Crawford 1994, May et al. 1997, Morley & Karr 2002, Tong & Chen 2002, Usseglio-Polatera & Beisel 2002, Wang et al. 2001). Urbanization is the suggested cause of the disappearance of anadromous fishes from tributaries of the Hudson River (Limburg & Schmidt 1990). Change in the amount of connected impervious surface was the best single predictor of fish density, diversity, and biotic integrity across a gradient from predominantly agriculture to predominantly urban land in southeastern Wisconsin (Wang et al. 2001). Increasing urbanization among 30 sites within the Etowah Catchment, Georgia, was negatively correlated with water quality, habitat, and measures of the macroinvertebrate assemblage (Roy et al. 2003). Despite the many factors thought to potentially limit Pacific salmon populations, percentage of urban land, along with water quality and sediment flow events, explained more than 60% of the variation in Chinook salmon recruitment in the interior Columbia River Basin from 1980–1990 (Regetz 2003).

Major changes associated with increased urban land area include increases in the amounts and variety of pollutants in runoff, more erratic hydrology owing to increased impervious surface area and runoff conveyance, increased water temperatures owing to loss of riparian vegetation and warming of surface runoff on

exposed surfaces, and reduction in channel and habitat structure owing to sediment inputs, bank destabilization, channelization, and restricted interactions between the river and its land margin (Table 1) (Paul & Meyer 2001).

Enhanced runoff from impervious surfaces and stormwater conveyance systems can degrade streams and displace organisms simply because of greater frequency and intensity of floods, erosion of streambeds, and displacement of sediments (Lenat & Crawford 1994). Modeled runoff within the Little Miami Basin, Ohio, was estimated to be more than 55 times greater from impervious than from pervious surfaces (Tong & Chen 2002). A comparison of streams in metropolitan areas and surrounding lands of Melbourne, Australia, found that macroinvertebrate taxa richness declined with increasing impervious surface, but streams of comparable imperviousness were markedly more degraded in the metropolitan drainage system. Flashiness of runoff was considered the primary influence throughout, but the presence of stormwater conveyance systems in the metropolitan area had the added effect of even greater flashiness and the conveyance of multiple pollutants (Walsh et al. 2001).

Whether urban land or impervious surface is a better predictor of the response of stream biota may depend on whether its primary influence is via flow alteration or also involves pollutants. Indeed, biological response measures have been better predicted by impervious area in several landscape studies of stream urbanization (Ourso & Frenzel 2003, Walsh et al. 2001, Wang et al. 2001) and by urban land area in others (Morley & Karr 2002), suggesting that hydrologic influences are primary in some studies, but the broader range of influences represented by urban area may be more important in others.

Because multiple pollutants enter urban streams, the direct influence of particular chemicals and metals is rarely demonstrated in comparisons of urban land use within catchments. Along a steep gradient of urbanization in the vicinity of Anchorage, Alaska, measured as a percentage of impervious area, macroinvertebrate taxa richness declined, and tolerant taxa replaced intolerant taxa (Ourso & Frenzel 2003). Urban land use, chemical factors, channel condition, and instream habitat all correlated with impervious area. However, stream and riparian habitat did not vary as strongly with impervious area as did water and sediment chemistry, suggesting that contaminants may have been of primary importance.

As a cause of changes in the biota, habitat degradation in response to catchment urbanization is less emphasized than other factors, particularly flow variability, although changes to bed sediments are commonly reported (Morley & Karr 2002, Roy et al. 2003). This deemphasis of habitat influence may be because some urban streams have protected corridors that maintain physical habitat but not water quality, or because habitat is relatively uniformly degraded among urbanized sites. Although catchment impervious area was the best single predictor of fish density, diversity, and biotic integrity in southeastern Wisconsin, stream habitat was not well correlated to increasing urbanization, which the authors attributed to prior habitat degradation associated with agriculture (Wang et al. 2001).

FOUR CHALLENGES

Undoubtedly, by changing the landscapes of stream catchments, human activities alter stream ecosystem in multiple ways (Table 1). However, our understanding of the relationships between anthropogenic land use and the ecological integrity of streams is complicated by covariation between anthropogenic and natural gradients, issues of scale, and uncertainties concerning the importance of legacies and thresholds. These challenges are now examined individually, although all may be of importance in a particular catchment study.

Covariation of Anthropogenic and Natural Landscape Features

Gradients of anthropogenic land use are frequently superimposed on an underlying gradient in parent geological material, soil type, topography, and other features of the natural terrain. Anthropogenic and natural factors covary because the latter influences the suitability of locations for agricultural and urban development. Sites near one another tend to be alike in both natural features and human uses, and spatial dependency can be anticipated in the distribution of organisms owing to their habitat requirements and tendency to disperse outwards from locations of high population recruitment (Corkum 1999). Whenever anthropogenic and natural gradients covary and only anthropogenic land use is assessed, the influence attributed to land use can be overestimated.

Whether natural or anthropogenic variables are found to have the stronger effect on stream condition depends substantially on the scope of the study, as well as on the adequate measurement of both types of variables. Nutrient and sediment measures often show that land use overrides natural features, particularly in agricultural lands (Johnson et al. 1997). In Lapwai Creek, an agriculturally impaired stream in northern Idaho, functional groups of macroinvertebrates were similar among sites despite expectations of differences along a river continuum, and the assemblage composition was markedly different from that found in less-impaired streams (Delong & Brusven 1998). Despite substantial variation in terrain and the extent of riparian vegetation, the relative homogeneity of the macroinvertebrate assemblages of these sites was interpreted, via increased sedimentation and the dominance of periphyton as an energy source, as evidence of the overwhelming effect of agricultural land use.

Natural factors may be of primary importance when human influence is minor, or when human influence is widespread and fairly uniform across the study region. In a study of 70 catchments within the relatively undegraded Northern Lakes and Forest Ecoregion of Wisconsin and Michigan, anthropogenic land use was not an important predictor of stream fish assemblages and attributes (Wang et al. 2003). The diversity and abundance of mollusks in the rivers of 36 catchments in Iowa were correlated with landscape factors indicative of erosional and groundwater processes, principally the average percentage of slope and percentage of land area composed of alluvial deposits (Arbuckle & Downing 2002). Possibly because the

entire area is highly agricultural, anthropogenic land use was not a predictor. The distribution of mollusks throughout a catchment in southeastern Michigan was well correlated with such local habitat measures as flow stability and substrate, which in turn were more strongly related to geology than to land use (McRae et al. 2004). Although geology and anthropogenic land use both varied throughout the catchment, the former appeared to have the stronger influence over local habitat conditions.

Coequal or at least mixed influence of natural and anthropogenic variables on local stream condition is a frequent finding. Both were influential in explaining patterns in macroinvertebrate assemblages among 55 riffle sites dispersed throughout the Taieri Basin of the South Island of New Zealand (Townsend et al. 2003). Overall, the distribution and abundance of macroinvertebrates were best explained by geomorphological factors at the catchment scale, by a mixture of geomorphology and land use at the reach scale, and predominantly by land use at the bedform (local, riffle) scale. Physical habitat variables explained approximately one third to one half of variation in macroinvertebrate assemblages, depending on season, among 46 sites in the Saginaw Basin of Michigan (Richards et al. 1996). Approximately one half of the habitat variation was explained by landscape variables. Bankfull width and other channel shape measures were much more strongly influenced by geology variables and only minimally by land use. Woody debris showed the opposite partitioning, whereas bankfull depth and canopy cover were equally influenced.

Covariation among natural and anthropogenic environmental factors can make attributing relative influence difficult or impossible. For example, a study of 25 agricultural streams in eastern Wisconsin found that agriculture was a strong influence, but the interpretation was complex owing to covariance of natural and anthropogenic variables measured at multiple spatial scales (Fitzpatrick et al. 2001). Streams with more than 10% agricultural land in their buffers were almost invariably impaired, particularly as indicated by an IBI for fishes, whereas invertebrate and algal metrics were less sensitive to land use. However, because riparian vegetation, geologic conditions, and hydrologic conditions were all correlated with the response of biotic metrics to agricultural land in the catchment, and because the relationships varied with the taxonomic group assessed, researchers could not confidently separate the interrelated effects of geologic setting, catchment and buffer land cover, and base flow.

Spatial Scale

Multiscale investigations often evaluate the relationship between stream condition and land use measured at several of the following scales: (*a*) the local reach, described by a buffer of 100 m to several hundred meters in width on each bank, and some hundreds of meters to a kilometer in length; (*b*) a buffer of similar width but of greater length, often the entire upstream distance for a small stream; and (*c*) the entire catchment upstream of a site (Figure 3). These scales will be referred

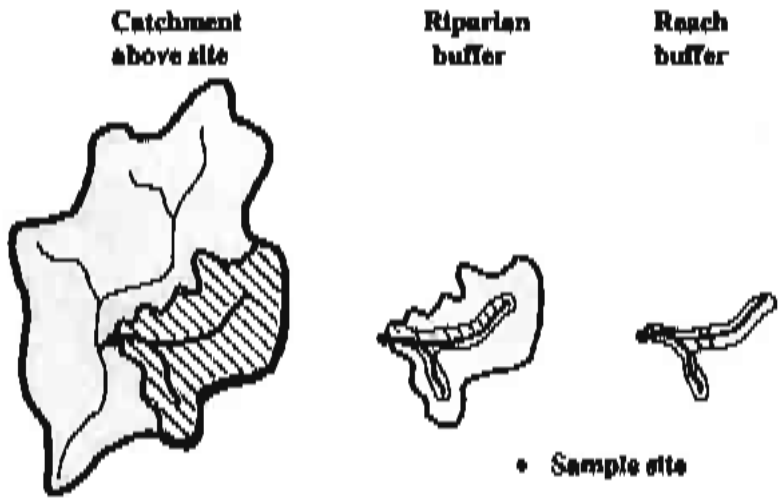

Figure 3 Three spatial scales widely used in relating landscape variables to some physical or biological measure of stream condition. The catchment typically is a subcatchment of a larger basin. Buffer widths of 100–200 m (each bank) are common. Modified from Morely & Karr (2002).

to as reach, riparian, and catchment, respectively: the former clearly is local scale, whereas the latter are two aspects of larger scale.

Different environmental variables of streams can be expected to vary in their responsiveness to large- versus local-scale environmental factors. Shade is influenced by very local patterns of riparian vegetation, water temperature responds to shading over distances of hundreds to thousands of meters, and inputs of leaf litter and wood are local but subject to downstream transport of variable distance (Allan et al. 1997, Quinn 2000). Nutrients and sediments can be transported long distances and so may be influenced by riparian conditions along a stream's entire length. Land use throughout the entire catchment governs stream hydrology (through its influence over evapotranspiration, infiltration, and runoff conveyance, and land use is a strong predictor of total nutrient loading (Boyer et al. 2002). The spatial scale at which an effect is detected is influenced by how closely land use in the riparian mirrors land use throughout the catchment, by data resolution, by the interplay of anthropogenic and natural gradients, and by specifics of study design. For example, a comparison of small catchments in southeastern Michigan that spanned a large gradient in agriculture but included only minimal replication of study reaches within catchments found that variation in land use at the catchment scale is the best predictor of stream habitat and fish IBI (Roth et al. 1996). However, reach-scale variation in land use was superior to catchment-scale variation in predicting stream condition within the same river basin when the study design examined multiple reaches within just three small catchments that differed

moderately in land use (Lammert & Allan 1999). Variation in land cover is often greater at the reach and riparian scales than at the catchment scale, which likely contributes to the greater influence attributed to riparian land use in many studies (e.g., Stauffer et al. 2000).

When land use at the reach and riparian scales is reported to have a strong influence over stream condition, direct local pathways are usually apparent. Pasture streams with occasional wooded reaches show marked physical and biological changes over distances of less than one kilometer. Forested reaches typically have cooler temperatures, wider channels, fewer sediments, and greater diversity of invertebrates (Abell & Allan 2002, Storey & Cowley 1997, Sweeney 1993). Fish assemblages in Ecuadorian streams changed from dominance by insectivorous and omnivorous taxa in pools with near-stream forest cover to primarily periphyton-grazers in open canopy pools, indicating a direct food web linkage (Bojsen & Barriga 2002). Near-stream connected imperviousness had a stronger influence on fish assemblages than did comparable amounts of impervious surface located farther from the stream, apparently owing to increased severity and frequency of high-flow events and lowered baseflow (Wang et al. 2001).

Even modest riparian deforestation in highly forested catchments can result in degradation of stream habitat owing to sediment inputs. A comparison of two small catchments that were less than 3% nonforested with two that were 13% and 22% nonforested found the latter to have higher concentrations of suspended sediments, higher turbidity at baseflow, five to nine times greater bedload transport, and greater embeddedness (Sutherland et al. 2002). Deforested riparian strips greater than one kilometer in length were associated with more fine sediments and less habitat diversity in a southern Appalachian stream, even though the riparian strips were vegetated and all were located within a highly forested catchment (Jones et al. 1999). Streams with reduced forest cover exhibit declines in overall fish abundance and an increase in sediment-tolerant and invasive species at the expense of those that spawn in clean gravel (Sutherland et al. 2002).

A number of studies have attributed more influence to catchment than to local land use, although pathways of influence may not be as easily detected. A composite index of habitat quality was strongly related to catchment land use and showed progressively weaker associations with riparian and reach-scale land use in southeastern Michigan, where a fish IBI was also more strongly associated with land use throughout the riparian and subcatchment than with reach-scale riparian vegetation (Roth et al. 1996). Invertebrate metrics were better predicted by catchment than by local-scale urbanization in the Puget Sound lowlands of Washington State (Morley & Karr 2002). Catchment-scale influence may be greatest when the primary mechanism is flow instability, nutrients, or some other factor related to the entire landscape.

Studies that examine a variety of measures of stream conditions in relation to land use at multiple scales report, unsurprisingly, mixed influence (Fitzpatrick et al. 2001, Richards et al. 1996, Roth et al. 1996, Stewart et al. 2001). Macroinvertebrate indexes were strongly correlated with both catchment and riparian land cover

over a range of 5%–61% total urban area and 34%–95% forest area in 100-m buffers (Roy et al. 2003). However, macroinvertebrate indexes were even more strongly predicted by environmental factors quantified at the reach-scale, including variation in substrate size and ion concentrations. Because reach-scale conditions were also associated with catchment land cover, these results are consistent with the view that large-scale landscape factors affect the biota via their influence over local-scale physical conditions.

Nonlinearities

Stream condition almost invariably responds nonlinearly to a gradient of increasing urban land or impervious area (IA). A marked decline in species diversity and IBIs with increasing urbanization has been reported from streams in Wisconsin (around 8%–12% IA, Stepenuck et al. 2002, Wang et al. 2000), Delaware (8%–15% IA, Paul & Meyer 2001), Maryland (greater than 12% IA, Klein 1979), and Georgia (15% urban land, Roy et al. 2003). Additional studies (reviewed in Paul & Meyer 2001, Stepenuck et al. 2002) provide evidence of marked changes in discharge, bank and channel erosion, and biotic condition at greater than 10% imperviousness. Although considerable evidence supports a threshold in stream health in the range of 10%–20% IA or urban land, others disagree (Bledsoe & Watson 2001, Karr & Chu 2000), and the relationship is likely too complex for a single threshold to apply. Hydrologic response is influenced by a number of catchment and stream characteristics, including slope, storage, conveyance and connectivity, and channel form (Bledsoe & Watson 2001, Walsh et al. 2001). Also, the supply of contaminants in urban storm runoff may vary independently of impervious area. In contrast to the above studies, a comparison of 45 highly urbanized sites around Seattle, Washington, reported a highly linear decline in macroinvertebrate indexes with increasing urban land and impervious area across the entire gradient (~10%–60% IA, ~20%–90% urban land; Morley & Karr 2002).

Streams in agricultural catchments usually remain in good condition until the extent of agriculture is relatively high, more than 30%–50%. In previously forested catchments in New Zealand, a macroinvertebrate fauna typical of undamaged sites was retained and abundances enhanced by conversion of up to 30% of catchment area to pastoral land, but increases in agricultural land above 30% resulted in an increase in pollution tolerant forms, illustrating a subsidy-stress relationship (Quinn 2000, Quinn & Hickey 1990). In several studies of Wisconsin streams, agricultural land use had a strong effect only when it exceeded 50% of catchment area (Wang et al. 2003, Wang et al. 1997). The response of stream condition to extent of agriculture across 172 sites from 20 major river basins throughout the United States was quite variable, and at least some sites had good fish condition even if agriculture exceeded 50% (Meador & Goldstein 2003). A study of agricultural streams in Wisconsin found indications of a decline in a fish IBI at >30% agriculture in the catchment and >10%–20% agriculture in the buffer (Fitzpatrick et al. 2001); another study reported declines in habitat quality and a fish IBI only when agriculture reached about 50% of catchment area, and some sites maintained high IBI and

habitat scores at 80% agriculture (Wang et al. 1997). The wide range of responses reported from streams draining agricultural landscapes clearly indicates that extent of agriculture is not by itself sufficient to predict the strength of the response.

Legacy Effects

Legacy effects are the consequence of disturbances that continue to influence environmental conditions long after the initial appearance of the disturbance. Observing that the present day diversity of stream macroinvertebrates and fish in forested catchments of the Appalachians, which previously had been farmed, were more similar to streams from present day agricultural landscapes than from present-day primary forest, Harding et al. (1998) emphasized the importance of the "ghost of land use past." Interpretation of the influence of land use is further complicated when cycles of change occur, such as when agricultural land reverts to forest, or when change is sequential, as when forested land is first converted to agriculture and subsequently to urban land. The finding by Wang et al. (2001) that fish metrics but not habitat varied strongly along an urbanization gradient was interpreted as the legacy of similar habitat degradation at all sites under the common, prior influence of agriculture.

Geomorphological changes brought about by multiple human activities likely have produced lasting, complex, and often unappreciated changes in physical structure and hydrology of river systems. Landscape changes that occurred within a few decades of European settlement of New South Wales, Australia, including clearance of riparian and floodplain vegetation and draining of swamps, have fundamentally altered river structure throughout virtually the entire Bega catchment (Brierley et al. 1999). Extensive habitat transformation has resulted, including channel widening and infilling of pools in lowland sections and incision of headwater channels owing to more efficient downstream water conveyance and downstream export of sediments. Overall structural complexity has been reduced and lateral connectivity is largely lost in middle reaches but is now increased in the lowlands. Unusually, longitudinal connectivity is now greater than was likely true of the presettlement, more discontinuous system. Brierley et al. (1999) estimate that it will take thousands of years for the sediment-starved upper reaches to refill with sediments, while at the same time the oversupply of sand in the lower river is now trapped by exotic vegetation. The timescale of recovery from geomorphic channel alterations is especially long, particularly in comparison to changes in land use, and so stream habitat and channel shape may never reach equilibrium with ongoing development.

How much the channels of some large North American rivers have lost complexity has only recently been appreciated. For example, the Willamette River, Oregon, is estimated to have undergone a fourfold reduction in length of shoreline as its once expansive floodplain and backwaters have been confined to a narrower and simpler channel in response to snag removal, channel dredging, and the draining of its floodplain (Sedell & Froggatt 1984). Under the combined influence of reductions in sediment supply and construction of levees, the Cedar River, Washington,

has experienced a 35% decrease in channel width and 45% decrease in channel area (Perkins 1994). Many streams of the upper Midwest have been deprived of wood by past timber harvest and the removal of existing wood in the channel by log drives (Johnson et al. 2003), and headwater regions have been transformed by the removal of beaver and consequent reduction in dams and ponds (Naiman et al. 1988). Elevated concentrations of heavy metals in the water column and in sediments of streams owing to hard-rock mining since the late 1800s were deemed responsible for reduced abundances and diversity of native fishes in northern Idaho (Maret & Maccoy 2002). Legacy effects owing to prior land clearing, channel modifications, snag removal, mining, and perhaps other human actions clearly pose a major challenge to linking present-day land use with concurrent stream condition.

MANAGEMENT APPLICATIONS

The measurement of stream health and its response to a variety of environmental stressors, including land use, requires well-tested indicators of ecological integrity. Composite measures, such as the IBI and percent assemblage similarity, are very useful in detecting overall stream degradation, but because of their aggregated nature they may be less easily interpreted than the behavior of individual response variables (Watzin & McIntosh 1999). For management and restoration actions to be effective, we must diagnose cause as well as assess harm, which requires an improved understanding of the mechanisms through which land use impacts stream ecosystems. Studies are needed that examine the response of individual species, traits, and guilds and that better connect the chain of influence from land use to stream response via studies of mechanisms.

Greater interpretability of stressor response relationships is usually achieved when the response variables are tolerance groupings, feeding and reproductive guilds, traits, and individual taxa (e.g., Poff & Allan 1995, Townsend et al. 1997, Usseglio-Polatera et al. 2000b). In comparing the responses of several macroinvertebrate metrics with several potential stressors using a large data set from the Mid-Atlantic Highlands, Yuan & Norton (2003) found the proportional abundance of tolerant taxa to be the most sensitive indicator of nutrient enrichment and habitat degradation, whereas Ephemeroptera richness was the most sensitive indicator of high metals and ions. Both linear and nonlinear response relationships were common. Using 11 biological and 11 ecological traits for 472 invertebrate taxa from French rivers, Usseglio et al. (2000a) identified distinct ecological groupings on the basis of body size, reproductive habitat, food source, and feeding habits, an approach that appears to hold promise for bioassessment (Gayraud et al. 2003). Further examples include shifts in fish reproductive guilds in response to sediment inputs (Jones et al. 1999, Sutherland et al. 2002), changes in the relative abundance of species that feed on periphyton versus leaf litter in response to loss of riparian shade (Bojsen & Barriga 2002, Quinn 2000), and an association between taxa with multivoltine life cycles and small body sizes and the extent of shallow, slow water habitat (Richards et al. 1997). Thus, Poff's (1997) argument that a

combination of landscape and habitat filters, together with categorizing or ranking taxa by traits that determine their susceptibility to particular environmental conditions, holds much promise for a multiscale, mechanistic understanding of assemblage response to changing land use and other broad-scale disturbances.

Riparian management is particularly attractive because of the riparian zone's immediate and direct influences on stream condition via well-documented pathways (Gregory et al. 1991, Naiman & Decamps 1997) and because it promises benefits that are highly disproportionate to the land area required (Lowrance et al. 1997, Quinn et al. 2001). However, gaps in the riparian (Weller et al. 1998) as well as subsurface farm and storm drains bypass the riparian zone and diminish its effectiveness (Barton 1996, Osborne & Kovacic 1993). In addition, landscape change at the scale of entire catchments may have impacts too great for a riparian strip to moderate. Studies that evaluate the influence of landscape change across multiple spatial scales report that stream responses are complex and interacting and vary with location and landform setting. For example, nutrient concentrations often reflect catchment land use, whereas macroinvertebrate assemblages appear especially sensitive to a number of local habitat factors (Hunsaker & Levine 1995, Strayer et al. 2003).

Reversal of land use to a less developed state at the catchment scale is rarely practical, and so improvement of stream condition more often depends on best management practices (BMPs) and improvements in landscape management and design. Some of these activities are at the catchment scale, such as conservation tillage, reduced fertilizer application, and other agricultural BMPs, as well as efforts to minimize hydrologic changes by retaining natural flow paths and infiltration capacity. Other BMPs are more proximate to the stream, such as stormwater retention ponds, managed wetlands, livestock exclusion, and maintenance of an intact riparian corridor. Evaluations of BMP benefits to stream condition commonly report improvements in physical and chemical variables, including habitat, nutrients, sediments, and turbidity (Caruso 2000, D'Arcy & Frost 2001, Lowrance et al. 1997, Strand & Merritt 1999, Wissmar & Beschta 1998). However, studies that evaluate biological responses to BMPs at the scale of the catchment are rare. One such study reported improvements in stream chemistry and streambed sediments in response to buffer practices, but the response of biological metrics was indistinct (Nerbonne & Vondracek 2001; see also Sovell et al. 2000). However, installation of riparian BMPs resulted in improvements in habitat quality and fish abundance in a Wisconsin stream (Wang et al. 2002). More studies of this kind are needed to determine whether physical improvements in stream condition are also evident in the biota.

The ecosystem functions performed by stream riparian zones vary with landform and location, as does human activity within the riparian, and so a "one size fits all" approach to riparian management is unlikely to be effective (Quinn et al. 2001, Strayer et al. 2003). Instead, knowledge of geomorphic setting and the key functions or uses of the riparian that are considered of greatest value should guide riparian management decisions. For example, Lowrance et al. (1997) estimated the amount of sediments, nitrogen, and phosphorus that forested riparian buffers

would retain from runoff entering the Chesapeake Bay for each of its nine physiographic provinces. They took into consideration the differences owing to soils, slope, and hydrologic connectivity and predicted different removal efficiencies for different pollutants. Ultimately, BMPs are likely to be chosen on the basis of their demonstrated effectiveness in a particular landform and human setting and of how much society values the expected benefit to the stream ecosystem.

The demonstrated effectiveness of land use data in predicting many components of stream condition points to an expanding role for landscape analysis in catchment management (Gergel et al. 2002). At present, most current studies rely on static Geographic Information Systems (GIS) maps that may represent land cover some years displaced in time from stream condition measures. However, remotely sensed data are likely to become more widely used in the future, offering greater opportunity to synchronize the time frame of land cover and stream condition measurement and to develop new landscape indicators. One promising demonstration showed that stream chemistry, habitat, and stream fish indexes across multiple ecoregions of Nebraska, Kansas, and Missouri were correlated to various "greenness" metrics on the basis of the normalized difference vegetation index, an indicator of vegetation condition and physiological activity obtained from satellite or airborne sensors (Griffith et al. 2002). Although management to mitigate land use impacts on streams will require site-based analysis of interacting factors, detection of areas at risk and estimation of probable risk factors are important and complementary activities to site-based studies.

CONCLUSIONS

The rapidly expanding investigation of streams in the context of their catchments and landscapes clearly indicates that stream ecosystems are strongly affected by human actions across spatial scales. The impacts are numerous, both direct and indirect, and complex, owing to the various pathways by which land use influences streams and the interaction between anthropogenic gradients and the hierarchically structured influence of landform on local stream conditions. Not only does the valley rule the stream, as Hynes (1975) so aptly put it, but increasingly, human activities rule the valley. The extent of change in river health in response to future population growth and development can be anticipated from knowledge of the relationships between land use and stream condition and plausible alternative futures (Baker et al. 2004).

Our understanding of the pathways and mechanisms through which land use influences stream conditions is informed by the comparative and empirical approach that has been the focus of this review; yet, it can also be said that this knowledge at present is extremely limited, particularly for prescriptive management. Our limited understanding is due in part to the multiple effects of a particular change in land use and in part to the influence of local setting and underlying natural variation. Clearly, the influence of the surrounding landscape on a stream is manifest

across multiple spatial scales and is further complicated by legacies from prior human activities. Thus, landform apparently operates mainly at the larger scale of catchment and region through its influence over geology, climate, vegetation, and topography, whereas the influence of land use operates across all scales, depending on the response variable of concern. Whether threshold responses are widespread is uncertain, owing partly to the scatter that is common in empirical relationships between land use and stream response. However, impacts of urban land use are clearly experienced at considerably lower percentages of catchment area than is true for agricultural land use, and most studies report a nonlinear response of stream condition to increasing urbanization.

Integrative measures of stream condition, including IBIs and percent similarity measures, are particularly useful for assessing overall stream health because they integrate multiple influences. However, species traits, feeding and reproductive guilds, taxa of known tolerance to particular stressors, and other less-aggregated measures are likely to prove more useful in evaluating pathways and mechanisms (Poff 1997, Usseglio-Polatera et al. 2000a). It will be particularly useful to examine the response of these more sensitive indicators to various management practices intended to offset the harmful impacts of intensive land uses. To date, the majority of catchment scale studies has only indirectly indicated tradeoffs, as in the common finding that biological metrics are negatively associated with agricultural land in the catchment but positively associated with forested land in the riparian (Steedman 1988, Wang et al. 1997). Future studies that examine the response of more revealing measures such as trait and guild composition, within a two-dimensional matrix of varying land use and management practices, could bring new understanding to the influence of land use on stream condition.

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EXHIBIT 4

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FENS OF THE UNITED STATES: DISTRIBUTION, CHARACTERISTICS, AND SCIENTIFIC CONNECTION VERSUS LEGAL ISOLATION

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Abstract: The term fen has been variously used by peatland ecologists, ground-water hydrologists, and vegetation scientists. The common denominator among all types of fens is recognition of the importance of ground-water discharge, especially mineral-rich ground water, in determining fen hydrology, chemistry, and vegetation, in contrast to wetlands whose characteristics are determined primarily by precipitation or surface-water inputs. Thus, fens tend to occur where climate and hydrogeologic setting sustain flows to the plant-rooting zone of mineral-rich ground water. In the United States, these areas include the glaciated Midwest and Northeast, as well as portions of the Appalachian Mountains and mountainous West. Individually and collectively, fens are among the most floristically diverse of all wetland types, supporting a large number of rare and uncommon bryophytes and vascular plant species, as well as uncommon animals including mammals, reptiles, land snails, butterflies, skippers, and dragonflies. Several species listed under the federal Endangered Species Act inhabit or use fens. Fens also help maintain stream water quality through denitrification and phosphorus sorption. Few estimates of loss and current extent exist, but where estimates are available, they indicate extensive loss, fragmentation, and degradation. Cultural eutrophication threatens the biological and functional integrity of remaining fens because, along with mineral-rich water, low availability of nitrogen and phosphorus controls many of their distinctive characteristics. Because they occur where ground water discharges to the surface, fens are isolated from neither ground water nor surface water. However, the majority of fens develop in headwater areas and could be defined as "isolated" for jurisdictional purposes because of their distance from navigable-in-fact waters. If so defined, the critical roles that fens play in maintaining biological diversity and stream water quality are at risk regarding federal jurisdiction over "isolated waters" because of the 2001 U.S. Supreme Court ruling in the case of *Solid Waste Agency of Northern Cook County v. U.S. Army Corps of Engineers.*

Key Words: biodiversity, calcareous fen, Clean Water Act, ground water, marl fen, poor fen, rare and endangered species, rich fen, species richness, SWANCC decision

INTRODUCTION

[illegible]

fens within the United States; (b) discuss their functional role in the landscape, especially in terms of biological diversity and maintenance of surface-water quality; (c) describe the various ways in which fens are both isolated and connected to other waters, and how their functional characteristics depend on their degree of both isolation from and connection to other waters; (d) discuss the extent to which fens are integral to meeting the goals of the 1972 Clean Water Act, i.e., maintaining the biological, physical, and chemical integrity of waters of the U.S. (interstate waters, navigable waters and their tributaries, and adjacent waters); and (e) estimate the impact of the SWANCC decision on fens.

METHODS

No comprehensive atlas of the fens of the United States exists. Most information published in the peer-reviewed literature focuses on one or a few fens; we relied heavily on the few exceptions (Richardson and Gibbons 1993, Nekola 1994, Moorhead and Russell 1998, Olivero 2001, Amon et al. 2002). Some types of information needed to address the question of the effect of the SWANCC decision on fens (e.g., the proportion adjacent to navigable waters and their tributaries) are seldom available. Therefore, we used four methods to obtain data and information relevant to the issues posed in this paper. First, the Wetlands Management Network of The Nature Conservancy (TNC) sent a request to all of their contacts in TNC state and regional offices and state natural heritage programs for information on fen abundance, distribution, occurrences of uncommon and rare species, fen area, and fen location relative to other waters. We received responses from 12 of these offices: Alabama, Arizona, Colorado, Michigan, Montana, New Hampshire, New Jersey, New York, North Carolina, Oregon, Pennsylvania, and Wisconsin. Second, we relied on the published literature to obtain additional data on fen distribution, plant and animal species composition, hydrology and water chemistry, fen area, and the location of fens relative to other waters. Third, we contacted other scientists whom we knew to be knowledgeable about fens in the United States and asked for similar types of information. Jeff Nekola, Carol Thompson, James Amon, David Cooper, Jim Almendinger, and John Sanderson provided us with unpublished data. Finally, we used data on New York fens that we and our colleagues have been collecting for the past twelve years. These include colleagues at Cornell University (F. Robert Wesley, Kathy Bailey, Carmen Chapin), the College of Environmental Science and Forestry at the State University of New York, Syracuse (Don Leopold), Russell Sage College (Nancy Slack), the Central and Western New York Chapter of The Nature Conservancy (Sandy Bonnano), and the New York State Natural Heritage Program (Carol Reschke, Adele Olivero). Specific sources are indicated in relevant figures and tables.

In order to estimate the contribution of fens to the support of plant biodiversity, we used both the published literature and contacts with state natural heritage programs and the scientists we knew to be working in fens in various states. From these sources, we estimated the number of vascular and bryophyte species that occur in fens, the total vascular and bryophyte flora for the state, the number of uncommon and rare plant species that occur in fens, the total number of uncommon and rare plant and animal species that occur in the state, and the area in fen. We defined uncommon and rare species broadly as any species listed as threatened, endangered, or a candidate for listing either at the federal or state level, as well as species designated as "special concern" by some states and species considered uncommon enough to be officially tracked by natural heritage programs. This last category typically included species with state heritage ranks of S1 (5 or fewer occurrences) and S2 (6–20 occurrences or factors demonstrably making the species extremely vulnerable), although some states also include the S3 rank (21–100 occurrences).

We used a geographic information system (GIS) and the location of New York fens to estimate the proportion of fens that are adjacent to or near navigable waters under different jurisdictional definitions of the term "adjacent." Geographic coordinates of 78 exemplary fens were obtained from the New York Natural Heritage Program and our own data. We used these coordinates to create a statewide fen point coverage geographically depicting the centroid of known NY fens. A statewide hydrography layer was created using county hydrography, i.e., 1995 Topologically Integrated Geographic Encoding and Referencing (TIGER/Line) hydrography files detailing both perennial and intermittent water features digitally obtained from the Cornell University Geospatial Repository (CUGIR) (U.S. Department of Commerce, Bureau of the Census 1995). These water features included rivers, streams, and lakes mapped as within or adjacent to New York State borders (e.g., Lake Ontario, Lake Champlain). These files were the highest available resolution, UTM Zone 18, with a positional accuracy of 36.48m. All data concerning lineage, data quality, and attribute information are available at CUGIR. County coverages were mosaicked using ESRI ARCVIEW 3.2a, resulting in a single statewide hydrography coverage. This hydrography coverage was then buffered at 100, 250, and 500 meters (Figure 1), simulating three adjacency scenarios. Isolation status of exempla-

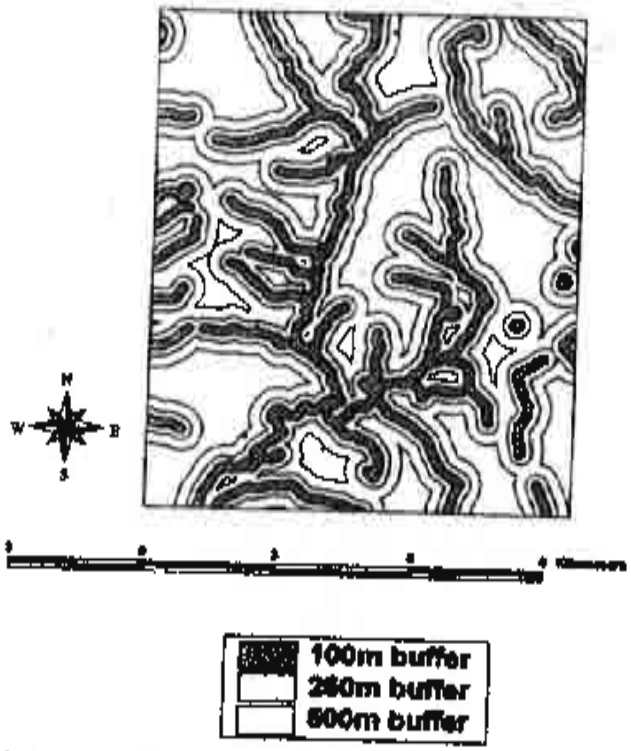

Figure 1. The location of ten exemplary fens of the Fall Creek watershed, Tompkins County, NY in relation to three adjacency scenarios (i.e., 100m, 250m, and 500m from waters mapped as permanent or intermittent on county-level hydrography maps) (see Methods).

ry fens was established by performing an intersection between hydrography buffers and fen point coverage, removing those fens that occurred within the buffer as being "connected" to waters of the United States. Thus, a fen could lie partially within the buffer but would be classified as "isolated" if its centroid was outside the buffer. We could not use a polygon coverage because more than a third of New York's fens are so small that they are not mapped on either National Wetland Inventory maps or New York's Department of Environmental Conservation jurisdictional maps. Because of the small size of New York fens, we think that the use of centroids is reasonable.

We also used a GIS to determine the distribution of fens in New York and Wisconsin in relationship to carbonate terranes. Digital maps at the scale of 1:250,000 of New York bedrock geology and surficial geology were acquired from CUGIR and used to create statewide coverages depicting fen relationship to carbonate mineralogy. Bedrock and surficial deposits were reclassified as containing carbonates (e.g., limestone, dolostone, calcareous kames) or not. We used the same geographic coordinates for fen location that we used to determine fen relationship to navigable waters. Digital images of Wisconsin geology (Wisconsin Geological and Natural History Survey 1981) and calcareous fen occurrence (obtained from the Wisconsin Department of Natural Resources and The Nature Conservancy of Wisconsin) were converted to TIFF image files and geospatially registered using IMAGEWARP (ESRI 2000) with an acceptable RMS error of 0.0025. Using these registered images, we then created an overlay depicting the approximate location of Wisconsin calcareous fens in relation to a generalized Wisconsin geology, with the shaded areas of this map corresponding to bedrock types containing calcium carbonates (e.g., dolomite, limestone, sandstone containing dolomite and limestone).

RESULTS

Definitions of Fens

Like many terms describing wetlands, the term fen has been used in various ways by peatland ecologists, hydrologists, botanists, and plant ecologists (Table 1). Specific uses include poor fen, rich fen, marl fen, sloping fen, calcareous fen, prairie fen, and extreme rich fen. The word itself has remained unchanged from the colloquial Old English *fen*, but its usage has evolved and continues to do so (Bridgham et al. 1996, Wheeler and Proctor 2000, Vitt 2000, Amon et al. 2002). Its scientific lineage can be traced back to the British ecologist, Arthur Tansley, who used the term in his 1911 and 1939 books on British vegetation to refer to a vegetation-soil complex characterized by alkaline soils and plant communities (Wheeler and Proctor 2000). According to Stuckey and Denny (1981), its first usage to describe alkaline wetlands in the United States was that of W.A. Anderson in a 1943 paper on an Iowa fen (Anderson 1943). Recent reviews of its usage and historical development can be found for peatland ecology in Bridgham et al. (1996), for European mires on both peat and mineral soils in Wheeler and Proctor (2000), and for midwestern U.S. fens in Amon et al. (2002).

The definitions in Table 1 clearly are not synonymous. Some emphasize water source; others emphasize water chemistry, vegetation, or type of substrate. These differences reflect both the scientific traditions and regions of the world from which they derive; hence, the definitions sometimes conflict. If origin of the water is emphasized, as in the older European tradition, then, depending on the mineralogy of the material through which ground water moves before it enters the fen, the chemistry of fens can be acid, circumneutral, or alkaline, with the vegetation reflecting these differences (Pearson and Leoschke 1992). The distinguishing factor is that fens are ground-water fed. If fen

Table 1. Definitions in current use of the term fen (unqualified) and some specific types of fen.

Fen (Midwestern U.S.)—a grassland on a wet and springy site, with an internal flow of water rich in calcium and magnesium bicarbonates and sometimes calcium and magnesium sulfates as well	Curtis 1959
Fen (Southeastern U.S.)—a minerotrophic ("groundwater-fed") wetland, richer in nutrients and less acid (pH > 4.8) than a bog and usually having a very slow internal drainage through seepage down low gradient slopes	Richardson and Gibbons 1993
Fen (Iowa)—wetland areas with saturated but not inundated soils which are fed by permanent groundwater seepage; sites which have saturated soils, but little or no peat accumulation, are considered fens	Nekola 1994
Fen (Canada)—peatlands that derive their water and nutrient supplies from precipitation and from water that has been in contact with upland soils	Zoltai and Vitt 1995
Fen (Midwestern U.S.)—wetland communities (1) dependent on ground water, which moves through and maintains saturation of the root zone throughout most of the year; (2) that do not experience long-term inundation; (3) that have carbon-accumulating substrates, including histosols ranging from fibrists to saprists, histic deposits, and carbonates such as marl and tufa; (4) that are dominated by non-emergent graminoid vegetation	Amon et al. 2002
Fen (England)—base-rich wetland with water pH values greater than about 5.5 but generally >6, with high calcium and bicarbonate, and vegetation rich in dicotyledonous herbs and brown mosses; a generic term for both herbaceous and wooded base-rich mires	Wheeler and Proctor 2000
Fen—somewhat less-acidic [than bog], more alkaline peatlands dominated by graminoids, brown mosses, taller shrubs, and coniferous and/or deciduous trees	Bridgham et al. 1996
Poor fen—an acidic, non-alkaline (pH 4.5–5.5) peatland dominated by *Sphagnum* mosses; influenced to some degree by water moving from surrounding uplands as well as precipitation; more similar to bogs than to rich fens in terms of vegetation and chemistry	Zoltai and Vitt 1995, Vitt 2000
Rich fen—fens having bicarbonate (thus they are alkaline) as their dominant anion and calcium as their dominant cation; pH > about 6.0; characterized by brown mosses largely of the family Amblystegiaceae, and an abundance of sedges; often used synonymously with calcareous fen	Zoltai and Vitt 1995, Vitt 2000
Calcareous fen—a term used to refer to rich fens with high calcium carbonate in water and soils, with peat or surficial deposits of calcium carbonate (marl) and a distinctive flora of rare calciphilic species, with pH in the range of 6.8–7.8; often used synonymously with rich fen	Eggers and Reed 1987, Komor 1994, Almendinger and Leete 1998b, Bedford et al. 2001
Marl fen—strongly minerotrophic wetlands in which the substrate is a marl bed derived either from lacustrine marl deposits or actively accumulating marl that is exposed at the ground surface, with pH generally higher than 7.5, and vegetation that is often sparse and stunted; because extreme-rich fens often accumulate marl, the two terms are sometimes used interchangeably	Olivero 2001

water chemistry is emphasized, as in the midwestern United States, then fens are distinguished from bogs in being alkaline or calcareous wetlands with pH above about 6.0 and in supporting calcicolous plant species. In this case, the source of the water also is ground water, but a wetland fed by ground water moving through sand or materials rich in iron sulphate, rather than through calcareous materials, would not be considered a fen because it would have a pH below about 5.5 and support vegetation with species that are calcifuges rather than calcicoles. On the other hand, the vegetation of these latter types of wetlands would not necessarily resemble that of bogs fed primarily by precipitation, nor that of poor fens associated with peatlands that receive some water from surrounding areas but nonetheless largely reflect the influence of precipitation in the plant rooting zone.

For purposes of this paper, we emphasize hydrogeological setting (HGS) (i.e., the physical setting of the wetland at the interface between ground water and surface water). We do so not because we disagree with other definitions, but because we find the ground water–surface water interactions that occur in these settings to be most relevant to the functioning of fens in the landscape, and hence to understanding the potential effects of the SWANCC decision on these wetlands and on meeting the goals of the Clean Water Act (1972). Thus, when we use the unqualified term fen, we mean wetlands that develop where a relatively constant supply of ground water to the plant rooting zone maintains saturated conditions most of the time and the water chemistry reflects the mineralogy of the surrounding and underlying soils and geological materials. These wetlands may be acidic to alkaline, have pH

ranging from 3.5 to 8.4 (Table 2), and may support calcicole or calcifuge plant species. They may occur on slopes, in depressions, or on flats (i.e., in a number of different hydrogeomorphic (HGM) classes) (Brinson 1993). Of course, like many factors structuring ecosystems, the degree to which ground water dominates fen water budgets is a continuum. In all cases used here, the influence of ground water exceeds that of precipitation and surface water, either in quantity or in terms of effects on water chemistry in the plant rooting zone.

Most basically, fens are ground-water-driven systems. Their hydrology, water and soil chemistry, and vegetation, as well as their function in the landscape, are determined in large part by the fact that they occur where ground water discharges to the plant rooting zone. Unlike bogs, whose rooting zones are dominated by inputs of atmospheric water, and marshes and swamps, which generally are strongly influenced by surface-water inputs, fen characteristics are determined by ground water. That is, ground water is a significant input term in fen water budgets, is responsible for maintaining saturation at the soil surface, or strongly influences water chemistry in the plant rooting zone. For example, Drexler et al. (1999a) estimated that ground water contributed 84–88 percent of the inflows to a small fen in central New York. Even the distinctions historically made among the different types of fen by boreal peatland ecologists (e.g., poor fen and rich fen) are a function of the extent to which ground water influences the plant rooting zone. Poor fens either (a) have poorly mineralized or poorly buffered ground water as a source, or (b) receive only a small proportion of their water from ground water and, hence, resemble bogs more than other fens in terms of vegetation and water chemistry (Gorham et al. 1984, Vitt 1994, 2000). Nonetheless, as Siegel (1983) showed, only a small amount of ground water input to a peatland can neutralize acid, raise pH above 5.0, and alter vegetation.

Because of relatively constant ground-water inputs, rich fens tend to have more stable water levels than other wetlands, particularly in temperate regions (Richardson and Gibbons 1993, Nekola 1994, Amon et al. 2002). Water is typically at the soil surface, with soils saturated but seldom flooded as they might be in a marsh, or several decimeters below the soil surface as it can be during droughts in marshes, bogs, and swamps. However, if the fen is fed by local rather than intermediate or regional ground-water flow systems, then the water table may drop during extended periods of drought, especially in arid or semi-arid regions. Based on studies in other systems and general hydrogeological principles, it is reasonable to assume that the larger and deeper the recharge area feeding the fen, the more stable the water table (Roulet 1990, Hill and Devito 1997).

Other distinguishing characteristics of fens also arise because of the influence of ground water. The most often reported of these is the base-rich nature of their water (Table 2) and soils (Slack et al. 1980, Vitt and Chee 1990, Anderson and Davis 1997, Amon et al. 2002), which is attributed to the movement of ground water through or over base-rich bedrock, glacial deposits, or soil before it enters the fen (Komor 1994, Almendinger and Leete 1998a). In most studies to date, these materials have been calcium and magnesium carbonates, such as limestone and dolostone, and ground water entering the fen carries a calcium-magnesium-bicarbonate signature (Komor 1994). However, rich fens also arise where ground water moving through material rich in calcium and magnesium sulfates (e.g., gypsum) discharges to the plant rooting zone (Curtis 1959, Almendinger and Leete 1998a, Godwin et al. 2002). Fens receiving high flows of calcium-rich water are called calcareous fens and support a distinct flora of plant species called calcicoles because of their affinity for calcium-rich sites (Almendinger and Leete 1998a and b, Lambers et al. 1998, Olivero 2001).

Depending on the flow rates and chemistry of ground water reaching the plant rooting zone, fens may be slightly acidic (poor fens), circumneutral (rich fens), or strongly alkaline (extreme rich fens, marl fens) (Table 2). Poor fens may arise either because ground water accounts for only a small fraction of their annual water budget or because ground water inputs move through materials with low solubility (e.g., gneiss, granite) or low buffering capacity (e.g., sand, quartz). Some of the fens of the Colorado Rockies (Cooper 1996, Johnson 1997) and the acid seeps or pine savannahs of New Jersey (New Jersey Natural Heritage Program, personal communication) fall in this latter category, as do the coastal plain poor fens of New York (Reschke 1990, Reschke et al. 1990). At the other end of the continuum, extreme rich fens and marl fens arise where carbonates precipitate at the fen surface. As ground water rich in calcium bicarbonate discharges to the surface at high rates, decreases in the partial pressure of CO_2 cause carbonates, such as calcite and calcium or magnesium carbonate (marl), to form in the plant rooting zone (Boyer and Wheeler 1989, Komor 1994, Futyma and Miller 2001). Rich fens lie in the middle of this continuum. In areas where ground water is supersaturated with respect to iron, iron hydroxides or iron sulfides may precipitate in the plant rooting zone. However, unless buffered by bicarbonate, these systems will have a low pH more typical of poor fens.

The accumulation of peat has been considered a key

Table 2. Water chemistry of North American Fens.

Source	Date	Location	Reported community	pH	Ca^{2+}	Mg^{2+}	Alkalinity
Sims et al.	1982	Ontario	Graminoid fen	6.6	21.9	4.1	
Sims et al.	1982	Ontario	Low shrub fen	6	16.6	5.4	
Sims et al.	1982	Ontario	Graminoid-rich treed fen	6.1	16.5	2.7	
Sims et al.	1982	Ontario	Sphagnum-rich treed fen	5.4	6.8	1.3	
Sjors	1963	Ontario	Fen	4.8	1.7	0.5	
Vitt et al.	1975	Alberta	Fen	5	2.4	0.4	
Heinselman	1970	Northern Minnesota	Fen	5.3–6.4	5–10.6	0.1–2.8	
Heinselman	1970	Northern Minnesota	String bog, patterned fen	5.0–6.6	4–10		
Schwintzer	1978	North Michigan	Fen	5.7–7.0	11–75	2.8–17.8	
Slack et al.	1980	West Alberta	Fen	6.8–7.9	4.18		
Siegel	1988	West Alberta	Ombrotrophic bog	3.8–4.1	0–5.7		
Siegel	1988	West Alberta	Rich fen	5.2–7.0	3.0–20		
Siegel	1988	Wels Alberta	Transitional	4.0–5.1	2.0–4		
Schwintzer and Tomberlin	1982	Lower Michigan	Bog	4.0 (0.2)	2.3 (0.8)	0.5 (0.2)	
Schwintzer and Tomberlin	1982	Lower Michigan	Conifer swamp	7.1 (0.2)	41.0 (7.6)	12.1 (1.6)	159 (25)
Schwintzer and Tomberlin	1982	Lower Michigan	Hardwood swamp	6.8 (0.2)	83.6 (30.5)	14.2 (2.1)	267 (97)
Schwintzer and Tomberlin	1982	Lower Michigan	Thicket swamp	6.9 (0.3)	54.7 (6.6)	6.2 (4.7)	213 (52)
Schwintzer and Tomberlin	1982	Lower Michigan	Fen	6.5 (0.4)	44.5 (21.9)	10.5 (4.9)	153 (74)
Schwintzer and Tomberlin	1982	Lower Michigan	Depression bogs	3.8–4.3	1.2–3.7		
Schwintzer and Tomberlin	1982	Lower Michigan	White Cedar swamp	6.7–7.4	28–50		129–197
Wilde and Randall	1951	Wisconsin	Spruce-tamarack muskeg	3.5–4.2			
Wilde and Randall	1951	Wisconsin	White Cedar swamp	6.2–7.3			24–134
Wilde and Randall	1951	Wisconsin	Sedge peat: marsh	6.4–7.0			144–332
Bellamy and Bellamy	1986	e. Canada	Minerotrophic mire	5.4–6.4	2.6–5.2	0.36–3.3	
Bellamy and Bellamy	1986	e. Canada	Transitional mire	4.1–4.5	1.2–3.4	0.24–3.3	
Foster and King	1984	Labrador	Patterned fens	4.7–5.5	0.3	0.2	
Foster et al.	1988	Quebec	Patterned fens	4.4–6.7	0.40–8.6	0.1–4.2	
Panno et al.	1999	Illinois	Fen	6.1–8.2	70–152	34–67	136–452
Stewart et al.	1993	Indiana	Fen	6.98–8.00	85–260	25–222	180–420
Almendinger and Leete	1998a	Minnesota	Fen	7.0–7.01	97–154	30–50	
Komor	1994	Minnesota	Fen	6.8–7.0	94–165	28–37	
Vitt and Horton	1990	Missouri	Fen	6.5–7.6			
Godwin et al.	2002	New York	Rich Fen	6.1–8.4	7–433	0.6–12	
Johnson and Leopold	1994	New York	Forested fen	6.9	31		
Johnson and Leopold	1994	New York	Open rich fen	6.7–6.9	27–31		
Johnson and Leopold	1994	New York	Poor fen	4.3–4.6	3–4		
Podnaesinski and Leopold	1998	New York	Forested fen	5.2–7.6	4–44		
Seischab	1984	New York	Rich and marl fen	7.5–7.9	58–60		
Western Pennsylvania Conservancy	1995	Pennsylvania	Seepage fen	6.2–7.9	3.0–105	0–39	
Motzkin	1994	western New England, eastern New York	Fen	6.0–8.1	8.0–65	4.0–32	

TABLE 1. [illegible]

Site	Ca	Mg	Na	K	Cl	SO4	Alkalinity
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]

[illegible]

characteristic of fens, usually within the peatland interior where poor and rich fens are viewed as part of the bog-rich fen continuum (Moore and Bellamy 1974, Wheeler and Proctor 2000). However, plant communities with the calcicolous species considered diagnostic of rich fens develop in mineral soils that receive ground water rich in calcium bicarbonate (Curtis 1959, Johnson and Largeski 1994, Van Hoewyk et al. 2000). The restriction of fens to peat probably derives from the fact that most literature on fens originates from boreal regions where peatlands dominate the landscape. In more temperate parts of North America, fens can develop on mineral, organic (peat), and carbonate substrates (Sutcliffe 1984, Sutcliffe and Brennan 1987, Sakaia 1994, Amon et al. 2002).

Scientists distinguish fens by their vegetation, particularly the presence of certain indicator species (Wheeler and Proctor 2000). Stuckey and Penny (1981), Zimmerman (1984), Eggers and Reed (1987), Pearson and Lemaitre (1987), and Minnesota Department of Natural Resources (1997) provide lists of vascular species considered indicative of rich fens. Vitt and Chee (1990) and Slack (1994) provide lists of bryophytes indicative of bog, poor fen, rich fen, extreme rich fen, and marl fens. Anderson et al. (1995) and Anderson et al. (1996) distinguish the bryophytes and sedges (*Cyperaceae*), respectively, of Maine bogs and fens. Amon et al. (2002) list vascular species typically found in fens of five midwestern U.S. states, and Oliver (2001) provides lists of bryophyte and vascular species associated to five specific types of calcareous fen in New York. Moorhead and Russell (1998) list bryophyte and vascular plant species commonly associated with mountain fens of the southern U.S. In general, the vegetation of fens is dominated by bryophytes, sedges (*Carex* and several other genera of the *Cyperaceae*), *Dicranelinum* herbs, and grasses. The vegetation of poor fens associated with peatlands more closely resembles that of bogs, with *Sphagnum* mosses and ericaceous shrubs dominant. Rich fens, however, are dominated by sedges and brown mosses (mostly of the Amblystegiaceae), with many distinctive species of dicotyledonous herbs. Although many definitions of fens restrict them to non-forested vegetation, the Canadian classification system (Zoltai and Vitt 1995) and recent discussions in the British literature (Wheeler and Proctor 2000) include forested wetlands receiving base rich ground water, as do Pedoniemi and Tarpold (1978) for New York and Crum (1988) in his classic text on peatlands.

One distinctive feature of fens, low availability of nitrogen (N) and phosphorus (P), is also associated with the fact that they are ground water driven systems. This characteristic has received appropriate emphasis only recently (Vitt and Chee 1990, Bridgham

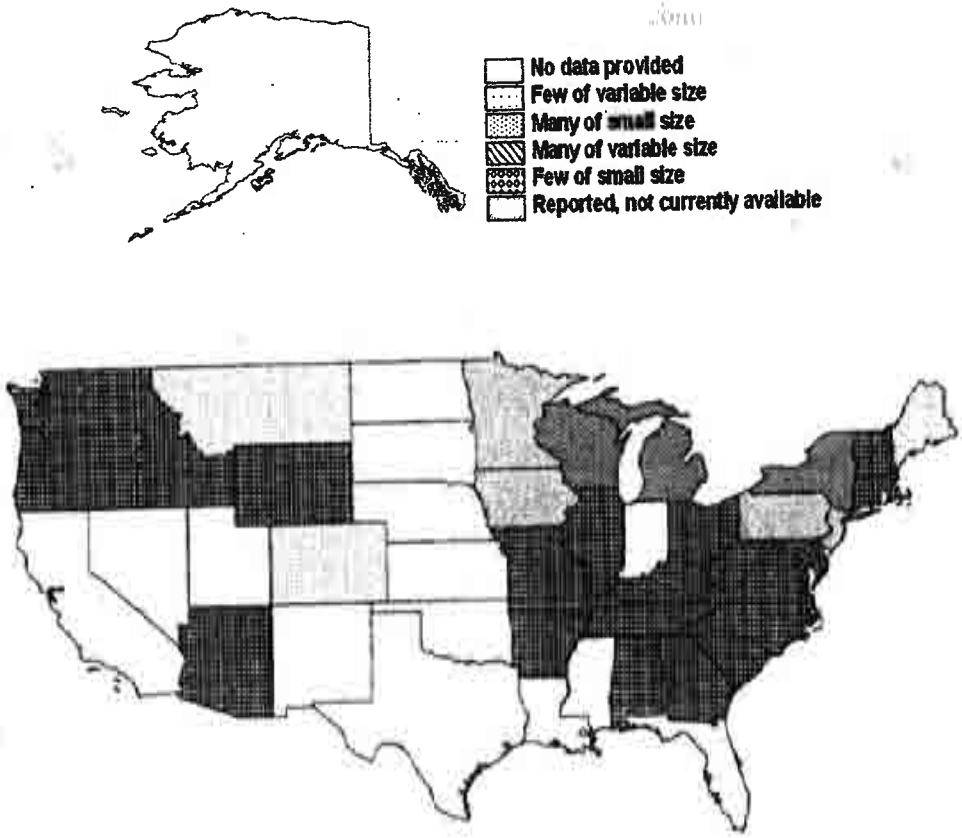

Figure 2. Fen occurrence in the coterminous United States and Alaska, by state, in relation to mean size and number of occurrences. Note that Alaska is not drawn to scale. "Not currently available" means that we were unable to obtain data on fen number or size. For sources, see Appendix A.

et al. 1996, Wheeler and Proctor 2000). The bog-poor fen-rich fen gradient, which correctly refers to a gradient in the number of indicator species and correlates with a gradient in base richness, has been confused with a gradient in the plant growth-limiting nutrients, N and P. In fact, the two gradients are more correctly viewed as independent and orthogonal (Vitt and Chee 1990, Wheeler and Proctor 2000). Unless enriched in N by atmospheric deposition, drainage from fertilized agricultural lands, or seepage from septic systems, fens are inherently low in available N. Phosphorus availability appears to be even lower, especially in rich fens. Rich fens typically have high N:P ratios in plant tissues, indicating P limitation to fen vegetation (Bedford et al. 1999), unless they receive run-off from urban or agricultural areas, or become subject to internal eutrophication through hydrologic alterations of fen water tables or water supply. The constant input of ground water rich in calcium bicarbonate, calcium sulphate, or iron ensures that most P in fens is adsorbed or precipitated into relatively unavailable forms of occluded calcium and iron-phosphate minerals (Stevenson 1986). The low stature, distinctive flora, and high plant species diversity of some rich fen vegetation may be attributable to these low nutrient conditions as much as to their base richness *per se* (Boyer and Wheeler 1989, Drexler and Bedford 2002).

Geographic Distribution and Hydrogeologic Setting

With the exception of poor fens associated with large peatlands, which occupy extensive areas of northern Minnesota, Wisconsin, Michigan, Maine, and Alaska, fens are a minor component of the U.S. landscape. They occur most commonly in formerly glaciated regions but also can be found south of the maximum extent of glaciation in Indiana and Pennsylvania, as well as in the mountains of Arizona, Appalachia, the Ozarks, Oregon Cascades, and Rocky Mountains. The occurrence of fens has been documented in 38 of the 50 United States (Figure 2, Appendices A and B). In most states, fens are few and small (a few to several ha).

As with other wetlands, the distribution of fens, as

well as the development of specific types of fens, is a function of climate and hydrogeologic setting (Winter 1992, Bedford 1996, Winter et al. 1998, Bedford 1999). Fens develop where ground-water discharge produces nearly continuously saturated conditions at the soil surface. This defining characteristic of fens is determined by (a) the climate of the region, which controls the balance between precipitation and evapotranspiration, and (b) several topographic and geologic features of the landscape that control the movement and chemistry of ground water: surface topography; slope of the land surface; stratigraphy and hydraulic properties of soils, surficial geologic deposits, and bedrock within the watershed and underlying the fen; mineralogy of the soils, surficial geologic deposits, and bedrock within the watershed and underlying the fen; the area of the wetland relative to the area of the watershed; and the flow-path length of ground water before it reaches the wetland (Winter 1988, 1992, Godwin et al. 2002). Thompson et al. (1992), Komor (1994), Almendinger and Leete (1998b), Amon et al. (2002), and the chapters by Carter and those for the states of Minnesota and Ohio in the U.S. Geological Survey's National Water Summary of Wetland Resources (1996) provide excellent illustrations of the major hydrogeologic settings in which fens occur.

These climatic and hydrogeologic settings appear to be rather restricted in the landscape. As Amon et al. (2002) point out, "in general, fens occur at stratigraphic and/or topographic breaks that create hydrologic gradients causing ground water to reach the land surface." In addition, climatic conditions must be such that saturation of the plant rooting zone is maintained nearly year-round. This means that fens develop only where recharge zones are sufficiently large and persistent or surficial deposits have high recharge rates and high transmissivities; topography and subsurface lithology force ground water to the surface; and climate and geology interact in such a way that evapotranspiration rates do not cause the plant rooting zone to dry out. Thus, in arid and semi-arid regions, and other parts of the country where evapotranspiration rates are high, fens form only where ground-water discharge is strong enough to offset evapotranspiration. Given such strong ties to the balance between ground-water inputs and losses from evapotranspiration, fens are likely to be susceptible to global climate warming and changes in ground-water flow due to irrigation or quarrying activities (Thompson et al. 1992, Winter et al. 2001).

The strong influence of geology on fen distribution is evident in the high spatial correlation between the location of specific types of fen and the distribution of particular types of bedrock and surficial geologic deposits. In New York, rich fens occur only where bedrock consists of limestone, dolostone, or marble, or

Figure 3. Distribution of 78 New York medium to rich fens with respect to geological materials. New York fens (circles) compared to the distribution of: (lower map) calcium-bearing bedrock, including limestone, dolostone, and marble (shaded areas); and (upper map) calcium-bearing surficial materials (shaded areas). See Methods for sources.

where surficial deposits of glacially-derived materials are calcareous (Figure 3). A similar correlation can be seen for Wisconsin (Figure 4), where calcareous fens are confined to areas where the bedrock consists of limestone, dolomite, or sandstone containing dolomite and limestone. Poor fens (not mapped) are restricted to areas where the bedrock is non-calcareous (e.g., sandstone, basalt, quartzite, granite).

Of the states with documented occurrences of fens (Appendix A), most report them to be few and small or many (greater than 100 occurrences) with most small. For example, there are fewer than 100 medium to rich fens in New York, and most are less than 5 ha (Figure 5)(Olivero 2001). The total area in fen for New Jersey, both calcareous fen and the non-calcareous, acid-seep pine savannahs, is about 200 ha (New Jersey Natural Heritage Program, personal communication).

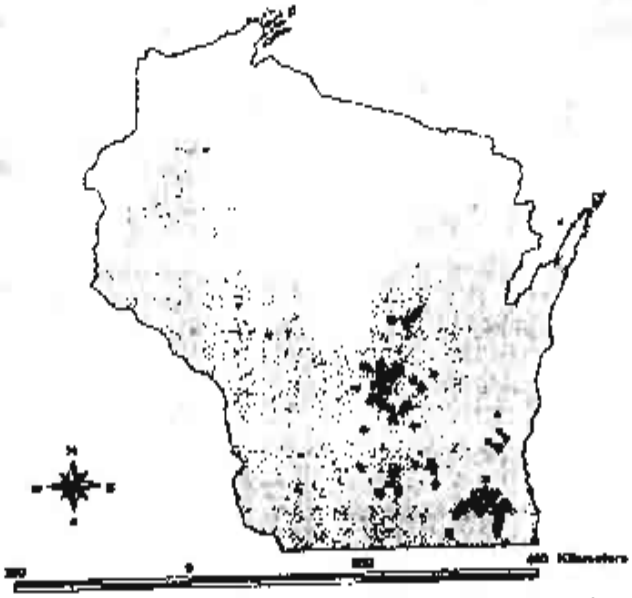

Figure 4. Distribution of 73 Wisconsin calcareous fens (circles) with respect to bedrock geology. Shaded areas correspond to the distribution of bedrock types containing calcium carbonates (e.g., dolomite, limestone, sandstone containing dolomite and limestone). Fen locations were provided by The Nature Conservancy of Wisconsin and the Wisconsin Department of Natural Resources. The map of bedrock geology is taken from the Wisconsin Geological and Natural History Survey (1981); map also available online at: http://www.uwex.edu/wgnhs/bdrk.htm.

North Carolina has a total of 140 ha in various types of fens and seeps (Weakley and Shafale 1994, North Carolina Natural Heritage Program, personal communication). The 23 documented calcareous fens in New Hampshire total only about 150 ha (Sperduto and Gilman 1995, New Hampshire Natural Heritage Inventory, personal communication). Nekola (1994) reported that fens now occupy only 0.01% of northeastern Iowa, and Cooper and Sanderson (personal communication) estimate that fens occupy 0.08–0.15% of Colorado. Wisconsin has 161 documented occurrences of fens, most of which are small (The Nature Conservancy of Wisconsin and Wisconsin Department of Natural Resources, personal communication). Only Michigan, Minnesota, Wisconsin, Maine, Alaska, and Colorado contain some large fens, primarily poor fens, although Colorado has one outstanding example of a large extreme-rich fen (Michael Kost, Michigan Natural Features Inventory, personal communication; James Almendinger, personal communication; The Nature Conservancy of Wisconsin and the Wisconsin Department of Natural Resources, personal communication; Anderson and Davis 1997). Montana has one large fen, the Pine Butte Fen, which is about 450 ha (Lesica 1986), but the other 62 documented occurrences are mostly less than 3 ha (Marc Jones, Montana Natural Heritage Program, personal communication; Chadde et al. 1998).

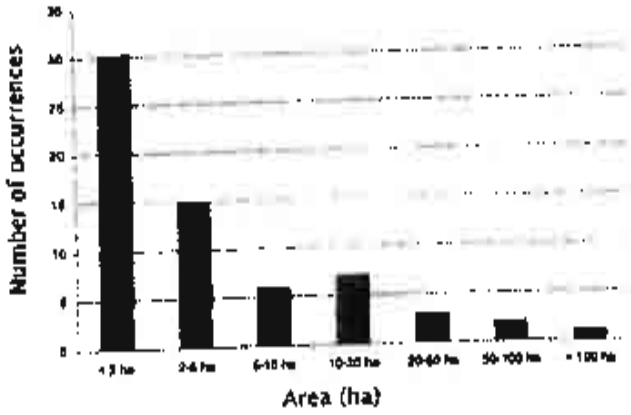

Figure 5. Area distribution of New York fens. Area of fens obtained from the New York Natural Heritage Program and our own data.

The current status of fens relative to their status at the time of European settlement is not well known. Fens were not distinguished from other wetlands in early accounts, and the current National Wetlands Inventory classification (*sensu* Cowardin et al. 1979) does not differentiate fens from other graminoid or shrub-dominated wetlands. An indication of the substantial loss of fens that likely has occurred since European settlement can be gained from a report for Iowa. According to a thorough inventory based on soils maps, topography, and site visits (Nekoa 1994), northeastern Iowa had 2333 fens as recently as 50 years ago, but only 160 remain extant. Pearson and Leoschke (1992) reported that nearly 40% of fen sites known to exist in Iowa in recent decades had been destroyed, that only about 12% had intact vegetation, and that most were very small, disturbed fragments. Many more fens, in Iowa and elsewhere, likely had been destroyed earlier in this century. Weakley and Shafale (1994) reported a six-fold loss of non-alluvial wetlands of North Carolina, all of which they describe as fed by ground-water seepage and, therefore, are fens. Of 52 fen sites known from Ohio in the latter half of this century, only 40 remained in 1981 (Stuckey and Denny 1981). Our observations in New York indicate that many remaining fens are degraded, either by beaver activity, by nitrate moving in ground water (Drexler and Bedford 2002), by phosphate adsorbed to sediment in run-off, or by altered hydrology and water chemistry caused by gravel mining and residential development within fen watersheds.

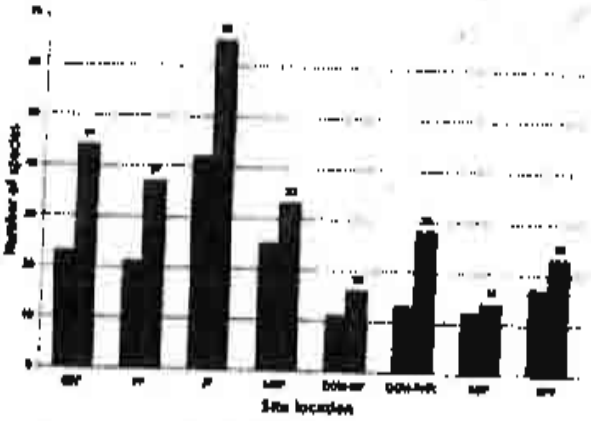

Figure 6. Plant species density in eight New York fens. The number of species per square meter (dark bars) is the mean of five plots with the standard deviation indicated by vertical lines. The number of species per five square meters is the total number of species in all five plots. The codes on the x-axis indicate individual sites with BSF, FF, JF, and MBF being rich fens, DCM-MF and DCM-PHR medium fens, RSF a poor fen, and SPF a medium fen.

Functions in the Landscape

Biodiversity. Wetlands that we now call fens, especially rich fens, have drawn the attention of botanists in the United States since the 1830s because of their diverse and unique flora (Stuckey and Denny 1981). In terms of within-plot species density, site-level species richness, regional diversity, and support of rare and endangered bryophytes and vascular plant species, as well as floristic oddities, fens are among the most diverse of all wetland types in the United States. Medium and poor fens in New York tend to have species densities in the range of 10–12 species per m², but the within-plot species density in rich fens can be as high as 42 species of vascular plants and bryophytes per m², with densities of 20–25 species being common (Figure 6). Few types of habitats world-wide support species densities this high. At the scale of 5 m², the species density of rich fens increases to between 33 and 65 species of vascular plants and bryophytes (Figure 6). Plots of 100 m² have densities of 20–80 species of bryophytes and vascular species (Reschke et al. 1990).

Because of small-scale environmental heterogeneity within sites, as well as biogeographical effects, fens tend to have high total species richness within sites (Stuckey and Denny 1991, Vitt et al. 1995). One of the most biologically diverse fens, Cedar Bog in Ohio, supports approximately 154 species of vascular plants in what Frederick (1974) described as the bog meadow (rich fen) and marl meadow, which occupy only about 8 hectares of the area she studied (extracted from Frederick 1974). In addition, at least 25 of the 92 species of mosses in Cedar Bog (Snider and Cusick 1989) appear to be rich fen species. According to Amon et al. (2002), the entire Cedar Bog preserve, which also includes swamp forest and arbor vitae swamp, contains 586 species in approximately 81 hectares. While Cedar Bog is undoubtedly an exceptional example because of its location at the interface of several biogeographical regions (Stuckey and Denny 1991), other fens also contain high site-level diversity. Lesica (1986) reported 93 vascular plant species and nine species of mosses for the 450 ha Pine Butte Fen in Western Montana. Nekola and Lammers (1989) recorded 100 species of vascular plants in a 10-ha Iowa spring fen. Drexler and Bedford (2002) list 26 bryophytes and 97 vascular species for a single 1.5 ha rich fen in central New York.

At larger spatial scales, those of landscapes and regions, fen plant diversity also is high, in part as a function of the different hydrogeologic settings within which fens occur in the landscape (Bedford 1999, Bedford et al. 2001, Godwin et al. 2002) and, hence, in environmental variability among sites. However, biogeography also plays a role at the regional scale (Stuckey and Denny 1991, Nekola 1999). Only 21 small (generally < 5 ha) fens in Pennsylvania contain 347 vascular plant species in 180 different genera and 70 families and 63 species of bryophytes in 37 genera and 23 families (Western Pennsylvania Conservancy 1995). Nekola (1994) reported a total of 320 vascular plant taxa in 160 small fens of northeastern Iowa. Chadde and Shelly (1995) reported 197 vascular plant species and 49 species of bryophytes in 19 western Montana poor and rich fens (see Chadde et al. 1998 for description of Montana peatlands). Olivero (2001) found 369 vascular species and 76 species of bryophytes in just 55 plots (100 m²) in 55 New York calcareous fens, undoubtedly less than the total New York fen flora. At a larger geographic scale, Amon et al. (2002) reported a total of 1126 vascular species in fens of eight midwestern U.S. states.

While the high plant diversity of U.S. fens is noteworthy, the numbers of uncommon, rare, threatened, and endangered plant species in these fens is even more so (Table 3). The characteristic that Nekola (1994) observed for Iowa fens, that they support a disproportionately high percentage of the state's rare flora relative to their area, seems to be true for several states. Fens, which occupy only 0.01% of the northeastern Iowa land area, harbor 12% of the state's total rare taxa and 17% of all Iowa "endangered," "threatened," and "special concern" species (Nekola 1994). Furthermore, Nekola (1994) found that Iowa fens support 18% of the total state flora. New York fens occupy about 0.07% of the state but support about 7% of its rare flora. New Jersey's calcareous fens and acidic seeps harbor 13.7% of the state's rare flora on less

Table 3. Contribution of fens to support of plant species diversity in selected states.

	Number of vascular species found in fens (# native)	Percent of state vascular flora found in fens (% native)	Number of non-vascular species found in fens	Number of uncommon & rare species found in fens	Percent of state uncommon & rare species found in fens	Percent of state area in fens
Colorado[1]	~500	~14		20	3.3	0.08–0.15
Idaho	327		20	35	12.0	
Iowa[2]	320	18		134	12.0	0.01
	(307)	(17.2)				
Montana[3]	174		60	40		0.0015
New Hampshire[4]	340	17.2	91	52	13.7	0.078
Calcareous fens						0.0058
Open fens						0.0726
New Jersey				96	13.5	0.00733
Calcareous fens	245			60		0.0073
Acidic seeps	169		13	36		0.00003
New York	440	13.8	77	55	7.0	0.07
	(397)	(19.0)				
North Carolina				77	11.0	0.0023

[1] For Colorado, the number of vascular species is an estimate provided by David Cooper and John Sanderson who have over 40 combined years of working in Colorado fens. The number of uncommon and rare species is taken from the Colorado Natural Heritage Program database by John Sanderson.
[2] For Iowa, the figures refer only to northeastern Iowa (29 counties), between 1/3 and 1/4 of the entire state.
[3] For Montana, we estimated the area in fen by adding the known area of Pine Butte Fen (450 ha) to the product of 61 (the number of documented fens in Montana minus the Pine Butte Fen) and 2 ha, a conservative estimate by staff of the Montana Natural Heritage Program of the area of the 61 fens. The number of bryophytes reported was obtained by subtracting the *Sphagnum* species not specifically mentioned by Chadde et al. (1998) as occurring in Montana fens from the total number of bryophytes listed by Chadde et al. (1998) as occurring in Montana and Idaho. Because they describe Idaho as rich in *Sphagnum* and Montana as rich in brown mosses, we think our estimate is reasonable.
[4] For New Hampshire, we report both the percent of the state (0.0058) that is calcareous fen (about 140 ha) and the percent (0.0726) that is open fen for the state (about 1745 ha), which includes poor level fen/bog, poor intermediate level fen, montane level-sloping fen, streamside fen, pond border fen, acidic patterned fen, one circumneutral-calcareous patterned fen (about 6.5 ha), and about 8.9 ha of calcareous fen. The estimate for the calcareous fens is considered a complete inventory but the estimate for the other types of fen is probably an underestimate (Sperduto and Nichols 2000). New Hampshire Natural Heritage Program, personal communication). However, the latter number is believed by the New Hampshire Natural Heritage Program to include all of the remaining high-quality fens in the state. Note that an additional 18 state rare species occur in "seeps," which would be considered fens by the definition we use in this paper, but for which we could obtain no estimate of area.
Sources: Colorado—D.J. Cooper, J. Sanderson, personal communication; Idaho—Chadde et al. 1998; Iowa—Nekola 1994; Montana—Montana Natural Heritage Program, personal communication, Chadde et al., New Hampshire—New Hampshire Natural Heritage Inventory 2002; New Jersey—New Jersey Natural Heritage program, personal communication, Breden et al. 2001; New York—Mitchell and Tucker 1997, Olivero 2001, New York Natural Heritage Program and F. Robert Wesley, personal communication; North Carolina—Murdock 1994, Moorhead and Rossell 1998, North Carolina Natural Heritage Program, personal communication.

than 0.01% of the state. New Hampshire, North Carolina, and Montana show similar patterns.

Fens within the United States also have been noted for their support of floristic oddities known as disjuncts (i.e., species far from their primary area of geographic distribution). For example, Colorado fens harbor several species whose main distribution is the arctic and boreal regions of Canada and the Great Lakes (David Cooper, personal communication; Hulten 1968 for distribution maps). These include *Salix myrtillifolia* Anderss., *Kobresia simpliciuscula* (Wahlenb.) Mack., *Scirpus (Trichophorum) pumilum* (Vahl.) Schinz and Thellung, *Carex livida* (Wahlenb.) Willd., *C. tenuiflora* Wahlenb., *C. pauciflora* Light., *C. diandra* Schrank., *C. scirpoidea* Michx., *Eriophorum gracile* Koch., *Potentilla palustris* (L.) Scop., and the mosses *Paludella squarrosa* (Hedw.) Brid., *Sphagnum balticum* Russ., *S. platyphyllum* (Lindb.) Braithw., and *S. girgensohnii* Russ. Nekola (1994, 1999) found over 80 species in Iowa fens to be disjunct from their distribution in boreal peatlands and those of northeastern North America. Curtis (1959), Frederick (1974), Stuckey and Denny (1981), Nekola and Lammers (1989), and Vitt and Horton (1990) all report the occurrence of disjuncts in the fen flora.

While the floristic uniqueness of fens has received the most attention, fens also support a large number of uncommon, rare, threatened, and endangered animals, including several federally listed species (Table 4). While the large mammals are wide-ranging and use fens only part of the time, several other species appear to use fens with a higher frequency than other habitats. These include the bog turtle (*Clemmys muhlenbergii* Schoepff) (Herman 1994 as cited in Moorhead and Rossell 1998) and eastern massasauga (*Sistrurus catenatus* Raf.). In addition to the federally listed species,

Table 4. Fauna reported to use or inhabit U.S. fens that are listed for protection under the federal Endangered Species Act. E = endangered; T = threatened; C = candidate for listing. (Compiled from U.S. Fish and Wildlife Service, Environmental Conservation Online Service [ECOS], last updated March 2002.)

Species	Federal Status	State(s)
Insects		
Icarice icarioides fenderi Boisduval (Fender's blue)	E	OR
Neonympha mitchellii French (Mitchell's satyr)	E	MI, OH, IN
Neonympha mitchellii franciscci French (Saint Francis' satyr)	E	NC
Somatochlora hineana Williamson (Hine's emerald dragonfly)	E	MI, IL, OH, WI, MS
Reptiles		
Clemmys muhlenbergii Schoepff (Bog turtle)	T	NY, OH, CT, MD, MA, NJ, PA, VI, NC, GA TN, VA, SC
Nerodia erythrogaster neglecta Conant (Copperbellied water snake)	T	OH, MI, IN
Sistrurus catenatus Rafinesque (Eastern massauga)	C	MI, NY, OH
Mammals		
Canis lupus L. (Gray wolf)	E	MI, WI, CO, MT, SD, WY, AZ, ID, WA, ND, MN
Canis rufus Audubon and Bachman (red wolf)	E	NC, TN
Felis concolor cougar L. (Puma)	E	NY, IL, IN, IA, MS, OH, CT, ME, MD, MA, NH, NJ, PA, VT, VI, WV, NC, SC, TN, VA, MI, MO, KY
Lynx canadensis Kerr (Lynx)	E	MI, NY, MN, WI, ME, NH, VT, CO, WY, ID, OR, WA, MT
Myotis sodalis Miller and Allen (Indiana bat)	E	NY, IL, IN, IA, MS, OH, MD, NJ, PA, VT, VI, WV, KS, AR, NC, GA, VA, MO, KY, TN
Ursus arctos horribilis Braunbären (Grizzly bear)	T	CO, MT, ID, WA, WY
Zapus hudsonius preblei Zimmermann (Preble's jumping mouse)	T	CO, WY

many state uncommon and rare species are found in fens. The state of Michigan lists at least one amphibian, four reptiles, and ten state rare insects as occurring in fens, in addition to the federally listed Mitchell's satyr (*Neonympha mitchellii* French), Hine's emerald dragonfly (*Somatochlora hineana* Williamson), and Eastern massasauga (*Sistrurus catenatus* Raf.) (The Nature Conservancy of Michigan, personal communication). Nekola (1994) reported that Iowa fens harbor 19 state rare butterflies and skippers. Murdock (1994) reported six species of rare, threatened, or endangered animals in the small mountain fens of North Carolina. These include the southern bog lemming (*Synaptomys cooperi* Baird), mole salamander (*Ambystoma talpoideum* Holbrook), and four-toed salamander (*Hemidactylium scutatum* Temminck and Schlegel).

Among the most interesting and least observed of animals unique to fens are the tiny land snails associated with calcareous fens. According to Jeff Nekola (personal communication), who has been studying snails in fens for the past decade, one of the rarest species in eastern North America, *Vertigo morsei* Strekl, is essentially limited to fens and has been recorded from only about two dozen sites. The species is much better known as a Pleistocene fossil. Other snails restricted almost entirely to fens include *Strobilops affinis* Pilsbry, which is limited to a narrow band ranging from Iowa to New Jersey, and two potentially new species, one a *Punctum* and one a *Helicodiscus* (Nekola, personal communication). Iowa fens harbor seven species of rare snails (Frest 1990 as cited in Nekola 1999). To date, Nekola (personal communication) has identified about 42 different species of land snails in the fens of Iowa, Wisconsin, Minnesota, and New York.

Surface-Water and Ground-Water Interactions. The hydrogeologic settings in which fens occur, always where ground water discharges to the surface, guarantee their strong influence on the physical and chemical properties of surface water. However, with the exception of Richardson and Marshall's (1986) study of the effects of fens on phosphorus retention, few studies from the United States have reported the effects of fens on downstream waters. The small size and striking biological diversity of many fens likely have turned attention away from effects on water chemistry. Nonetheless, we can draw several conclusions from current

understanding of nitrogen transformation and phosphorus retention in wetlands and studies that have reported fen soil and water chemistry.

Fundamental principles indicate that wetlands with high amounts of calcium, iron, or aluminum in their waters and soils have a high potential to retain phosphorus, with aluminum and iron being more important at low pH and calcium being more important at high pH (Richardson 1985, Richardson and Marshall 1986, Stevenson 1986, Walbridge and Struthers 1993). Adsorption and precipitation of phosphorus with ions of these elements effectively move phosphorus from available to unavailable pools (Boyer and Wheeler 1989), although phosphorus sorbed to iron is subject to redox changes associated with water-level changes (Darke and Walbridge 2000). Calcium concentrations in fen waters can be quite high, notably in rich fens, calcareous fens, and marl fens (Table 2), all of which have the high pH conditions under which phosphorus is removed from available pools. Some fens, however, also can have high concentrations of iron that bind phosphorus. New Jersey's acidic seeps (pine savannahs) have so much iron in discharging ground water that "bog iron" (i.e., goethite, an iron oxyhydroxide) was once harvested from them (Joan Ehrenfeld, personal communication). Fens in New York have iron concentrations in shallow ground water ranging from below detection (0.05 mg/L) to 0.46 mg/L. New York's rich fens have soils in which iron-bound P accounts for about 11% of the total soil phosphorus pool (mean = 11.1%, s.d. = 2.6, n = 35) (Chapin and Bedford, unpublished data).

Fundamental principles also indicate that wetlands have a high potential for denitrification (Groffman 1994). While rates of denitrification have been found to be low in both bogs and fens in Europe (Waughman 1980, Koerselman et al. 1989, Verhoeven et al. 1996), these low rates are a function of the lack of a nitrate supply in undisturbed peatlands. However, if there is a supply of nitrate, as from surrounding agricultural lands, grazing animals, or atmospheric deposition, then the anoxic conditions and high carbon content of fen soils can be expected to promote high rates of denitrification, thus removing nitrate from surface water before it reaches downstream rivers and lakes. Groffman et al. (1996) reported variable soil denitrification enzyme activity in New York fens, with soils from a single fen showing higher activity than soils from three other types of wetlands. Van Hoewyk et al. (2000) later reported that denitrification enzyme activity, on a per gram basis, was greater in fens on organic soils than those on mineral soils. Furthermore, net nitrogen mineralization in fens can be negative (i.e., the fen removes more nitrogen from the available pool than it releases, leaving none to be transported to downstream waters) (Groffman et al. 1996, Van Hoewyk et al. 2000). Drexler et al. (1999a) found that ground water entering a New York fen at a spring below a corn field had nitrate concentrations of 7.8 (± 2.9) mg per liter, while stream outflows had concentrations of 1.6 (±0.8) mg per liter.

With ground-water discharge as a defining characteristic, fens modulate the temperature of waters flowing from them because ground water tends to maintain a rather constant temperature reflecting the mean annual air temperature of a region. In general, fen waters and soils are cooler in summer and warmer in winter than air temperatures and other surface waters in the region (Amon et al. 2002). Frederick (1974) took detailed records of soil in Ohio's Cedar Bog (rich fen) and compared them to a nearby agricultural experiment station. She found that soil temperatures in the bog meadow (rich fen) never went below freezing and that the average maximum temperature for the six warmest months was 8.8° C cooler in the bog meadow (rich fen) than in upland soils at the experiment station. She also found that Cedar Run, the stream emanating from the fen, never froze and observed that in winter "Cedar Run appeared to "steam" because of the great differences in air and water temperatures" (p. 71). The Mad River, to which the Cedar Bog and Cedar Run feed, has long been recognized for its sustained high flow of clear, cold water (Frederick 1974). Almendinger and Leete (1998a) reported that samples drawn from water-table wells in three eastern Minnesota calcareous fens had a mean summer temperature of 11.1° C, which varied only 3.3° C, while mean monthly air temperatures for June, July, and August are 20.1° C, 23.1° C, and 21.4° C, respectively (State Climatology Office, Minnesota Department of Natural Resources, http://climate.umn.edu). Stewart et al. (1993) reported a mean surface water temperature for nine Indiana fens of 14.3° C (s.d. = 3.6, n = 27).

Relationship to Other Waters: Connection and Isolation

In a landscape context, the functional characteristics and significance of fens depend oddly on both their isolation from and connection to other waters. While their biological diversity is controlled in large part by their connection to ground-water flows and to the chemistry of ground water (Almendinger and Leete 1998b, Drexler et al. 1999b), it also depends in part on a kind of isolation from other surface waters (i.e., on an environment that is usually saturated by ground water but seldom flooded by nearby surface waters) (Stewart et al. 1993, Amon et al. 2002). Were fens regularly flooded, several physical and chemical properties that govern their vegetation would change. Nu-

trient and sediment loads would be greater, thus promoting production of greater biomass and probably reducing species diversity (Grace 1999). Regular flooding also would eliminate many plant species intolerant of such conditions, thus further reducing species diversity. Many of the species that occur in fens simply do not occur in wetlands with standing water. Some of the rare and endangered animal species also would not occur with regular flooding and higher sediment and nutrient loads. Bog turtles, the massasauga, and the land snails appear to prefer wetlands, such as fens, with little standing water (Peter Rosenbaum, personal communication, Jeff Nekola, personal communication). In this restricted sense, fens, while intimately "connected" to ground-water flow, are "isolated" from other surface waters. While they may feed other surface waters, the flow is almost always unidirectional.

On the other hand, the biodiversity of fens may be threatened by their increasing isolation in the landscape (Nekola 1999). Except for large expanses of poor fen in peatlands of Maine, Minnesota, Michigan, and Wisconsin, fens likely have always been isolated patches within a larger matrix of other types of ecosystems because of the particular combination of hydrologic, geologic, and stratigraphic factors required for their development. However, as human activities have destroyed fens, the distances between fens have increased. For example, Nekola (1999) mapped the distributions of presettlement and extant fens in northeastern Iowa. Along with the expected increase in geographic isolation of the fens from each other, he found that isolation was a significant predictor of plant species richness, explaining between 46% and 84 % of the variance in species richness accounted for by fen area. He attributed this to lowered species-immigration rates between sites relative to presettlement levels and the lowered probabilities of species being able to reestablish themselves following local extinction events. Thus, assuming similar size, fens more geographically isolated from each other are likely to have lower species richness than those close together. Furthermore, with time, the effect is likely to increase, further lowering species richness.

From the standpoint of maintaining water quality and flows to streams and lakes, the intimate connection of fens to surface waters is fundamental. Their position at the interface between ground water and surface water means that they are a primary buffer between downstream waters and nutrients and other pollutants moving from upland areas. Obviously, the cool summer temperatures and the ionic signature of surrounding glacial deposits of streams flowing from fens is a function of their connection to these ground-water discharge sites.

Relationship to the Goals of the Clean Water Act

As ecological systems that occur where ground water becomes surface water, fens play a critical role in maintaining the physical, chemical, and biological integrity of U.S. waters, the primary goal of the 1972 federal Clean Water Act. They are particularly noteworthy for their (a) exceptional support of plant and animal biodiversity; (b) their position at the headwaters of streams and lakes, and hence as major influences on stream and lake chemistry; and (c) their buffering of surface-water temperatures by supplying water that is cooler in summer and warmer in winter than other surface waters.

Potential Impacts of the SWANCC Decision on U.S. Fens

The potential impacts of the SWANCC decision on fens of the United States are difficult to estimate for reasons pertaining both to our knowledge of the distribution of fens in the landscape and to ambiguity in how the Supreme Court's decision is to be interpreted (Downing et al. 2003). As a recent summary by the National Wildlife Federation (Goldman-Carter 2002) of post-SWANCC court decisions makes clear, the federal and district courts are not interpreting the decision, nor the terms associated with it, uniformly. The differences largely reflect narrow versus expansive interpretations of the SWANCC decision and how the terms "navigable," "tributary," and "adjacent" are defined. Under a narrow reading, the Supreme Court eliminated only the use by migratory birds to establish a connection to navigable waters and left open the question of whether or not means other than migratory birds could be invoked to establish a connection to navigable waters. In this case, if a surface-water connection could be traced from a fen through a tributary and then to a navigable water, the fen might fall within federal jurisdiction. Examples include cases in which the courts held intermittent tributaries to navigable waters to be jurisdictional, declared a creek located 378 km from the nearest navigable-in-fact water to be jurisdictional, and upheld federal jurisdiction for a wetland connected to navigable waters by an intermittent surface connection (Goldman-Carter 2002). In contrast, if an expansive interpretation of the SWANCC decision is applied, then only those fens actually adjacent to navigable-in-fact waters would be covered by Clean Water Act protection. Examples include a case in which a continuous surface connection to a navigable water of 32 linear km was held to be too far to exert jurisdiction, and a case in which only bodies of water that are actually navigable or adjacent to navigable waters were held subject to Clean Water Act

Table 5. Relationship of 78 exemplary New York fens to navigable waters. Buffer distance refers to the width of the band around all New York waters mapped as permanent or intermittent on county-level hydrography maps.

Buffer Distance	Number of Fens Within Buffer
100 meters	21
250 meters	34
500 meters	51

Note: Only 26 of the 78 fens included are located in wetlands mapped by the New York State Department of Environmental Conservation as jurisdictional wetlands (i.e., subject to New York State wetland regulations).

jurisdiction (Goldman-Carter 2002). Thus, only fens that are themselves navigable, which is never the case, or are directly adjacent to traditionally navigable waters, or lie on the boundary of two states (i.e., are interstate fens), would receive protection.

Our knowledge of the spatial distribution of fens does not allow us to estimate directly what proportion of U.S. fens are directly adjacent to navigable waters, which have a continuous surface-water connection with navigable waters but are in fact distant from them, or which have no surface-water connection to navigable-in-fact waters. The National Wetlands Inventory does not differentiate fens from other palustrine wetlands. Many fens are too small even to show up on topographic maps or state jurisdictional wetland maps or be differentiated from other ecosystems based on high-altitude aerial photographs.

We, therefore, used two approaches to estimating the impact of the SWANCC decision on fens. The first of these investigated the proportion of fens that would be jurisdictionally defined as isolated at three discrete distances from the surface waters of New York. In this analysis, we found that only 21 of 78 exemplary New York fens are located within 100 meters of a navigable body of water (Table 5). Even if we stretch the notion of "adjacency" or "proximity" to 500 meters, 27 of the 78 fens are not within the buffer. The analysis is likely to be similar for other glaciated eastern states where fens are located in similar climatic and landscape settings. Thus, under an expansive interpretation of SWANCC, most New York fens, and probably most eastern U.S. fens, would not be considered to be under Clean Water Act jurisdiction even though a continuous surface water connection to navigable water probably could be traced in all cases.

Our second approach was to seek the expert opinion of scientists, regulatory staff, and naturalists who know the fens of their states. Their insights indicate that, under a narrow interpretation of the SWANCC decision, most fens in the humid and mountainous regions would remain under Clean Water Act jurisdiction because a continuous surface water connection usually could be traced to navigable waters, albeit often at great distances (Table 6). On the other hand, fens located in semi-arid regions are not likely to be either adjacent to nor continuous with navigable waters. Fens in humid-continental regions may or may not be adjacent to or continuous with navigable waters. For example, most of Iowa's fens, where the climate is humid-continental and streams outflowing from fens often dry before reaching major streams, would be excluded from jurisdiction. Under an expansive interpretation, most fens of the U.S. would no longer be subject to Clean Water Act regulations because few are actually adjacent to traditionally defined navigable waters (Table 6). Although all of New Jersey's acidic seeps are adjacent to streams, the total area of these fens is only about 57.5 ha.

CONCLUSIONS

The most striking characteristic of U.S. fens is their disproportionately large contribution to the biological, chemical, and physical integrity of the Nation's waters

Table 6. Expert opinion on the relationship of fens in various states to navigable bodies of water.

State	Continuous with but Distant from Navigable Waters	Adjacent to Navigable Waters	Neither Continuous with nor Adjacent to Navigable Waters
Colorado	most	probably none	small percent
Iowa	few	few	most
Michigan	~half	many	14–20
Minnesota	most	<5 percent	small percent
New Jersey			
calcareous fens	almost all	<25 ha	probably none
acidic seeps	few	almost all	none
New York	most	few	few
Montana	most	few	few
North Carolina	most	few	~20 percent

relative to the small part of the landscape they occupy. With the exception of the boreal poor fens of Maine, Michigan, Wisconsin, and Minnesota, nowhere do fens form more than even one percent of the landscape. Nonetheless, because they sit at a critical interface—in headwater positions where ground water becomes surface water—they strongly influence the flows, chemistry, and temperature of water entering our Nation's streams and lakes. Their plant species diversity is unequaled among wetland ecosystems and high relative to all other U.S. ecosystems. They also support many species of uncommon and rare animals, notably several federally listed species and many species of state-listed rare butterflies, skippers, moths, and land snails. Additionally, fens expand the range of many regionally rare and endangered species. If the SWANCC decision is interpreted narrowly, most fens in humid regions would not be affected, but fens in semi-arid regions would no longer be covered under the Clean Water Act. In contrast, if the decision is interpreted expansively, then most fens in most regions would be considered "isolated" for jurisdictional purposes and not subject to Clean Water Act regulation. In this case, the legal definition and scientific understanding of the concept of isolation clearly would be at odds.

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Appendix A. Documentation for Fen Occurrences in the United States.
Notes: The list of references is not inclusive. See Nekola 1994, Amon et al. 2002, and references therein and the references given here for additional references. Richardson and Gibbons (1993) and Moorhead and Rossell (1998) use different terms for the same wetlands, all of which would be fens by the definition used in this paper. p.c. = personal communication.

State	Types of Fens Described	Source
Alabama	mountain bogs, Appalachian plateau fens	Richardson and Gibbons 1993, Moorhead and Rossell 1998
Alaska	floating mat fens, calcareous fens	Racine and Walters 1994, McClellan et al. 2003
Arizona	poor fen	The Nature Conservancy, Arizona, p.c.
Arkansas	sedge-shrub fen, acid seep forest, Ozark plateau fens, mountain seeps	Richardson and Gibbons 1993, Moorhead and Rossell 1998
Colorado	extreme rich fen, treed fen, moderately acid fen,	Cooper 1996, Cooper and Sanderson 1997, Johnson 1997, Cooper et al. 1998
	circumneutral fen, subalpine treed fen	Johnson 1997, Cooper et al. 1998
Connecticut	calcareous fens	Motzkin 1994
Georgia	mountain bogs (poor fens), valley & ridge fens, Blue Ridge fens	Richardson and Gibbons 1993, Moorhead and Rossell 1998
Idaho	valley and sub-alpine peatlands	Chadde et al. 1998
Illinois	shrub fen, graminoid fen, calcareous seep	Nekola 1994, Stoynoff and Heat 1986
Indiana	alkaline hard-water fens	Wilcox et al. 1986, Stewart et al. 1993, Nekola 1994, Swinehard 1997
Iowa (northeastern)	herbaceous fens fed by permanent groundwater seepage, spring fen	Nekola and Lammers 1989, Nekola 1994
Iowa (statewide)	fens, herbaceous seepage-based fens, peatlands fed by shallow groundwater	van der Valk 1975, Pearson and Leoschke 1992, Thompson and Bettis 1994
Kansas	fens	Nekola 1994
Kentucky	mountain bogs, Appalachian plateau fens	Richardson and Gibbons 1993, Moorhead and Rossell 1998
Maine	rich fens, basin & valley unpatterned fens, closed-basin fens, ribbed fens	Jacobson et al. 1991, Anderson and Davis 1997
Maryland	fen	Shaw and Rabenjorst 1997
Massachusetts	calcareous fens	http://www.NatureServe.org, Motzkin 1994
Michigan	fen	Schwintzer 1978, Schwintzer and Tomberlin 1982
Minnesota	poor fen, minerotrophic fen, calcareous fen	Glaser 1987, Komor 1994, Almendinger and Leete 1998a,b
Missouri	rich fen	Witt and Horton 1990, Nekola 1994
Montana	minerotrophic fen, rich fens, intermediate fen, poor fen	Lesica 1986, Chadde et al. 1998
Nebraska	fen	Nekola 1994
New Hampshire	poor fen, intermediate fen, sloping fen, streamside fen, pond border fen, circumneutral calcareous fens, calcareous fen	Sperduto and Gilman 1995, Sperduto and Nichols 2000, Sperduto et al. 2000, New Hampshire Natural Heritage Program, p.c.
New Jersey	calcareous fens, acidic seeps	New Jersey Natural Heritage Program, p.c.
New York	marl pond shore, marl fen, rich sloping fen, rich graminoid fen, rich shrub fen, medium fen, inland poor fen, coastal plain poor fen	Reschke 1990, Godwin et al. 2002
North Carolina	mountain bogs, southern Appalachian fen	Richardson and Gibbons 1993, Murdock 1994, Weakley and Schafale 1994, Moorhead and Rossell 1998
North Dakota	fen	Duxbury et al. 1986, Malterer et al. 1987, Nekola 1994
Ohio	bog meadow, marl meadow, fen meadow, marl fen	Frederick 1974, Stuckey and Denny 1981
Oregon	rich fen	The Nature Conservancy of Oregon, p.c.
Pennsylvania	calcareous seeps, calcareous fens, acidic seepage swamps, calcareous swamps	Davis 1993, Western Pennsylvania Conservancy 1995
South Carolina	mountain bog, Blue Ridge fens	Richardson and Gibbons 1993, Moorhead and Rossell 1998
South Dakota	fen	Nekola 1994
Tennessee	mountain bog, Blue Ridge fen	Richardson and Gibbons 1993, Moorhead and Rossell 1998
Vermont	open sedge fen, sloping fen, poor fen, wooded fen, medium fen	Coles 1987, The Nature Conservancy of Vermont, p.c.

Appendix A. Continued.

State	Types of Fens Described	Source
Virginia	mountain bogs, valley and ridge fens	Richardson and Gibbons 1993, Moorhead and Rossell 1998
Washington	subalpine peatlands	Chadde et al. 1998
West Virginia	Marl fens, mountain bogs, Appalachian plateau fens	Bargis and Lang 1984, Richardson and Gibbons 1993, Moorhead and Rossell 1998
Wisconsin	alkaline fen, calcareous fen, boreal rich fen, central poor fen, poor fen, shore fen	Curtis 1959, Amon et al. 2002, The Nature Conservancy of Wisconsin, p.c.
Wyoming	calcareous peatland, transitional rich fens	Cooper and Andrus 1994, Chadde et al. 1998

Appendix B. Sources for Unpublished Data Used in This Paper

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EXHIBIT 5

9 Hydrological Processes, Nutrient Flows and Patterns of Fens and Bogs

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9.1 Introduction

The total area of northern peatland has been estimated to be 5.8×10^6 km^2 (Table 9.1) A great deal, particularly in Europe, has been drained for agriculture, forestry, housing and other human use. The ability of wet, pristine peatland (mires) to accumulate atmospheric carbon, to clean water and to buffer river floods and their values of characteristic biodiversity urges for conservation action and the restoration (rewetting) of drained peatland. Hydro-ecological field data from pristine mire ecosystems will enhance modeling approaches for the prediction of effective water and land use management aimed at mire conservation and restoration.

The scarce information on pristine mires leads to unjustified negative opinions and sometimes even biased scientific results. For an estimation of the world surface area of peatland, most publications refer to the data compiled by Gorham (1991). This leads to the effect that the importance of Russian peatland is strongly underestimated in climate change modeling, i.e. the depletion of atmospheric carbon by terrestrial sinks is much larger because the surface area of Russian peatland is about double the size of 1.5×10^6 km^2 given by Gorham (1991). Merely in Western Siberia, the surface area of peatlands is more than 1×10^6 km^2 (Yefremov and Yefremova 2001). Further, more than 90 % of the Russian peatland is still unaffected by draining activities and is sequestering atmospheric carbon.

Most of the bogs and fens in other regions have been drained by human activities aimed at lowering the groundwater table, extracting the groundwater for water supply and irrigation, regulating the river channel and discharge, etc. and can be designated today as former mires or peatland. In Europe and

Table 9.1 Surface area of peatlands in Northern America and Eurasia

Location	Surface area of peatlands (km²)	Reference
Asian Russia	3 121 000	Vompersky et al. (1999)
Europe	916 762	Bragg (2002)
United States of America	550 600	Gorham (1991)
Canada	1 270 000	Waddington et al. (1998)
Total northern peatland	5 857 262	

beyond, the wish to restore fens and bogs is growing because of the high biodiversity values; but because of the slow adaptation of vegetation to changed hydrological conditions, the level of degradation is difficult to establish. Therefore, predicting the effectiveness of rewetting aimed at nature restoration is hampered by a lack of knowledge of the pristine hydrological and hydrochemical (nutrient) conditions of fen ecosystems.

This chapter reports on empirical and modeling studies performed in relatively undisturbed mires in the northern hemisphere of America, Europe and Asia. First, the major mire vegetation types of Western Siberia are presented, next we go deeper into regional and local hydrological studies performed in bogs and fens and finally we summarize the results of an eco-hydrological analysis performed in the Ob valley.

9.2 Appearance of Pristine Fens and Bogs

9.2.1 General

Mires reveal close relations between topography, hydrology, hydrochemistry and vegetation (Kulczynski 1949; Moore and Bellamy 1973; Ratcliffe 1977; Gore 1983). In general, differences in the topographical and hydrological position of mires give rise to different types of mire water, especially concerning nutrient and calcium contents and acidity, which in turn determine the floristic composition and diversity of the mire vegetation (Sjörs 1950; Succow and Joosten 2001).

Peat formation is effective where plant litter decay is reduced by low redox potential, as happens in water-saturated conditions. Such conditions can develop in climates suitable for plant growth (trees, herbs, mosses, lichens) at places where precipitation exceeds evapotranspiration and which have a low soil infiltration capacity. Peat bog mires occur in particular at places with a

flat topography and large distances between rivers and drainage channels, where water-table mounds can develop. Such mounds favor the growth of raised bogs by: (1) driving local recharge (downward) flow systems that isolate the surface waters from the influx of inorganic solutes derived from mineral soil and (2) allowing the water-table to mound upward and saturate new layers of peat.

Mires fully dependent on nutrient supply by dry and wet (snow included) atmospheric deposition are typified as 'ombrotrophic'. Mires which in addition to atmospheric deposition receive dissolved nutrients and minerals by groundwater flow are mostly named 'minerotrophic' or 'rich' fens. Poor (oligotrophic) fens receive water and nutrients from the atmosphere and from adjacent ombrotrophic bogs. Starting from groundwater-fed mires, raised (domed) bogs can develop while losing contact with groundwater.

Big mire complexes present in Canada, United States and Siberia can include various mire types. Raised bogs, developed mostly at or near the water, divide the discharge surplus water to downstream stretches of large poor fen 'lawns', occasionally with teardrop-shaped bog islands. Further down and between the teardrop islands, patterned ridge-hollow (string-flark) bogs develop, with elongated hollows perpendicular to the water flow direction. Also, water tracks may be present in these poor fens. At locations where water tracks cut into the mineral subsoil, where the peat layer is thin or at the margins of mire complexes groundwater seeps up, feeding the minerotrophic fens.

9.2.2 Climate and Mire Vegetation of the Western Siberian Taiga

By its characteristics of vegetation and climate, the taiga zone of Western Siberia between the Ural mountains and the Yenisey river has been split into three main zones: Southern Taiga (56–58° N), Middle Taiga (58–61° N) and Northern Taiga (61–64° N). From south to north, precipitation increases (0.51–0.57 m year^{-1}) and there is a decrease in both radiation (0.32–0.25 kCal m^{-2}) and summer temperature (average July: 18–14 °C).

As a result of this climate gradient, higher evapotranspiration rates facilitate slightly higher average ion concentrations in the Southern Taiga compared to the Northern Taiga. The electroconductivity of peat water diminishes from 5–12 mS m^{-1} in the south to 3–7 mS m^{-1} in the north, mainly caused by a change in hydrocarbonate content. Nitrate and orthophosphate contents are generally lower in the north (0.4–0.1 g NO_3^- m^{-3}, 1.1–0.2 g PO_4^{3-} m^{-3}); but only for the bog type 'ryam' (Table 9.2) nitrate decreases exponentially with latitude (r^2=0.77). Nutrient concentrations were explained better by different mire types then by climatic zones (Bleuten and Lapshina 2001).

Table 9.2 gives the vegetation, species and species diversity of the major mire types of the Southern Taiga, ordered along increasing nutrient availabil-

Table 9.2 Total number of species by layer (tree/herb/moss/lichens) and dominant plant species of main mire types of Western Siberian taiga zones, based on 1278 Braun-Blanquet vegetation relevés

Mire type	Description	Species data	Species
1	Poor dwarf shrubs bogs ('ryam')	76 species (15/32/19/6)	*Ledum palustre*, *Chamaedaphne calyculata*, *Oxycoccus microcarpus*, *Rubus chamaemorus*, *Drosera rotundifolia*, *Eriophorum vaginatum*, *Sphagnum fuscum*, *S. magellanicum*, *S. angustifolium*
2	Hollows and *Sphagnum* lawns	33 species (13/6/14)	*Carex limosa*, *Scheuchzeria palustris*, *Eriophorum vaginatum*, *E. russeolum*, *Drosera rotundifolia*, *D. anglica*, *Rhynchospora alba*, *Sphagnum balticum*, *S. papillosum*, *S. lindbergii*, *S. jensenii*, *S. majus*, *Cladopodiella fluitans*
3	Poor through-flow fens and water tracks		
3.1a	Poor sedge *Sphagnum* moss fens	43 species (–/16/27)	*Carex rostrata*, *Chamaedaphne calyculata*, *Oxycoccus palustris*, *Menyanthes trifoliata*, *Drosera anglica*, *Sphagnum fallax*
3.1b	Poor sedge–herb *Sphagnum* moss fens	106 species (–/49/57)	*Carex lasiocarpa*, *Comarum palustre*, *Lysimachia thyrsiflora*, *Galium trifidum*, *Calliergon stramineum*, *Sphagnum obtusum*, *S. centrale*, *S. teres*, *S. subsecundum*, *S. platyphyllum*, *S. contortum*, *Schistichylopsis laxa*
3.2	Poor sedge brown moss fens	89 species (–/56/33)	*Carex lasiocarpa*, *C. elata* ssp. *omskiana*, *C. chordorrhiza*, *C. heleonastes*, *Dryocolaris intermedia*, *Juncus stygius*, *Scorpidium scorpioides*, *Warnstorfia exannulata*, *W. fluitans*, *Meesia triquetra*, *Campylium polygamum*, *Pseudocalliergon trifarium*
4	Rich sedge brown moss fens	97 species (–/30/67)	*Carex diandra*, *Thelypteris palustris*, *Saxifraga hirculus*, *Stellaria crassifolia*, *Epipactis palustris*, *Dactylorhiza incarnata*, *Parnassia palustris*, *Drepanocladus sendtneri*, *D. adunca*, *Hamatocaulis vernicosus*, *Bryum pseudotriquetrum*, *Tomentypnum nitens*
5	Wooded swamps ('nogra')	373 species (16/153/204/6)	*Carex elongata*, *C. lolinea*, *C. disperma*, *Dryopteris cristata*, *Cardamine trifida*, *Listera cordata*, *L. ovata*, *Malaxis monophyllos*, et al., many liverworts, as well as *Mnium stellare*, *Thuidium recognitum*, *T. philibertii*, *Timmia megapolitana*, *Rhizomnium punctatum*, *R. pseudopunctatum*
6	Tussock-sedge flood plain mires	153 species (–/68/85)	*Carex cespitosa*, *C. juncella*, *Salix cinerea*, *S. rosmarinifolia*, *Spiraea salicifolia*, *Calamagrostis purpurea*, *C. phragmitoides*, *Lathyrus palustris*, *Veronica longifolia*, *Lysimachia vulgaris*, *Scutellaria galericulata*

ity (from ombrotrophic to minerotrophic). Bog mires of *Sphagnum*–dwarf shrub communities with stunted (0.5–4.0 m) pines, in Russian language called 'ryam' (type 1), cover large areas and are present as ridges in patterned bog complexes, together with hollows and lakes. Ryam-type present in the Southern Taiga resembles the permafrost types like flat palsas and ridges of patterned bogs of the Northern Taiga. Actual vegetation differs by the low coverage of lichens in the Southern Taiga (ca. 6 %) compared to the Middle Taiga (14 %) and the Northern Taiga zone (69–85 %). Ryam types with higher trees are missing in the North. Hollows and *Sphagnum* lawns (type 2) are limited to ombrotrophic conditions of waterlogged plains and some parts of old river terraces. The species diversity of hollow vegetation types, which sometimes cover large waterlogged areas on the flat top of waterlogged mires, appear to be very low: only 33 species including mosses and lichens have been recorded.

Through-flow fens (type 3) receive water from oligotrophic bog complexes and from 'teardrop' ryam bog islands. Water flows very slowly over the upper peat layer and through the living plant and moss layer of these types. Locally, this water converges into water tracks. Two sub-types and two variants have been discerned.

The first variant (variant sub-type 3.1a) consists of oligotrophic sedge–*Sphagnum* moss of through flow fens within large ombrotrophic peatland. The second variant (variant sub-type 3.1b) comprises slightly minerotrophic sedge–herb–*Sphagnum* moss fens, also called transitional mires, developed on the periphery of large watershed mires and along the uplands of river terraces.

Poor sedge–brown moss fens (sub-type 3.2) are widespread in the relatively drier climate parts of the Southern Taiga zone watershed plain between the Ob and Irtysh rivers area.

Very wet rich brown moss fens (type 4) fed by carbonate-rich groundwater have developed in non-flooded parts of river valleys and on lower river terraces. Rich brown moss fens, as well as poor brown moss through-flow fens are only present in the Southern Taiga and sub-Taiga zone (transition between Southern Taiga zone and Steppe zone), caused by a subsoil of carbonate-rich loess sediments. Probably the very low net precipitation may cause more exchange with deeper groundwater, as also was reported from the Glacial Lake Agassiz Peatland study (Glaser et al. 2004a, b).

A wide variety of woodland fens on shallow to thick (up to 6 m) peat, common in Western Siberia is represented in this classification by only one type (type 5). These fens are mostly fed by groundwater, are characterized by a high species richness and include forests of tall trees. In the nutrient-rich flooded parts of the river plain, extensive mires (type 6) with dense shrubs and sometimes high (up to 1 m) tussocks of sedges are present.

9.3 Hydrology of Bogs: Examples from Canada, United States and Western Siberia

9.3.1 Aspects of Large-Scale Hydrology

For peat accumulation, the soil surface should be water-saturated for a great part of the growing season. Such conditions can be found in shallow freshwater wetlands as lakes, marshes, river valleys and also on broad flat land of water divides. The Red Lake Peatland (1200 km², 48° N, 95° W) in the United States (Glaser et al. 1981) and the Great Vasyugan Bog Complex in Western Siberia (57 000 km², 56–58° N, 76–83° E) which both developed on a flat water divide area (Bleuten and Lapshina 2001) are examples of the latter type of areas. There, the classic sequence of mire types, as described above, from ombrotrophic bogs to minerotrophic fens, is present in a downhill sequence.

Deviations from these generally accepted relations between groundwater flow and raised bog formation have been found for the Hudson Bay Lowlands (HBL: 51° N, 83° W) along the Albany River (Canada) and the Glacial Lake Agassiz Peatland (GLAP: 48° N, 94° W) region of Minnesota (USA). The large bog mire complex of the Hudson Bay Lowlands (51° N, 83° W) along the Albany River (Canada) have been built up in an area the drainage of which was disturbed by post-glacial isostatic upheaval (Glaser et al. 2004a, b). There, the rate of uplift increased spatially toward an uplift dome centered over the present mouth of James Bay. The emerging coastline of the lowlands has therefore been rising at a faster rate than regions of the interior throughout the post-glacial period, thereby: (1) decreasing the regional gradient, (2) raising the regional water table and (3) driving the formation and development of peatland. This hypothesis is supported by the synoptic analysis of vegetation, water chemistry, hydrogeology and peat stratigraphy of representative peat landform types across the study area, which straddles a peat-covered till plain and subtly mounded moraine (Glaser et al. 2004a, b). Broad fen water tracks, in contrast to a normal position, developed down-slope from the moraine complex and in the lower reaches of the local catchments on the till plain, where mineral solutes are transported upward through the peat cover by advective dispersion along lateral flow paths or groundwater discharge.

In the Glacial Lake Agassiz Peatland (48° N, 94° W) region of Minnesota (USA), however, peatland development was largely driven by climate-driven flow systems. The actual regional climate, however, is unexpectedly dry with only a slight moisture surplus and periodic droughts that perturb groundwater flow systems. Initial local and synoptic hydrogeologic surveys in this region showed that spring-fens and some fen water tracks were consistently located over discharge zones for groundwater, whereas most of the larger fen water tracks developed in areas of lateral flow. One unexpected result of this work was that the raised bogs also developed over discharge zones for

groundwater (Siegel 1983; Romanowicz et al. 1995; Siegel et al. 1995; Glaser et al. 1997). During moist periods, water-table mounds within these bogs drove surface waters downward and deflected the upwelling groundwater to the bog margins, but during droughts, the water-table mounds dissipated, permitting the groundwater to rise closer to the peat surface.

9.3.2 Local Scale Hydrology of Bogs

Water flow systems of bog mires include three main processes: (1) overland flow through the green part of the bog ('acrotelm') above the mean lowest groundwater level, (2) sub-surface shallow groundwater flow through the upper peat layers and (3) deeper groundwater flow. Of these three, overland flow is considered the most important. In order to test this hypothesis, an intensive measurement program monitoring pH and electrical conductivity (EC) has been completed in the Haukkasuo concentric bog (60°45' N, 26°57' E) in southeastern Finland (Mäkilä 1997). The monitor network included 710 measurements of temperature and EC in two perpendicular and vertical cross-sections (length: 1900 m and 1150 m). The distance between the probing sites was 100 m and the measurements were made from the bog surface to the bottom, at 0.25 m spacing. The longer transect was measured six times at different seasons. In four sites, the peat column was sampled down to the bottom by taking 89 volumetric samples with a special piston sampler.

Acidity

The acidity of the Haukkasuo bog decreases quite smoothly from pH 3.2–3.5 at the surface towards pH 5.4–6.0 at 6 m below the peat surface at the central parts of the bog. Near the bog margins, the acidity decrease is faster with depth: pH 6.0 is reached at 2.0–2.5 m below the peat surface. The source of acidity is located near the bog surface, where free hydrogen ions are produced by the cation exchange ability of *Sphagnum* moss and peat, and where organic acids are released by the aerobic decomposition of *Sphagnum* (Shotyk 1988). In addition, the bog surface receives acid precipitation water (pH=4.6). The highest pH values (pH>6.0) are reached near the mineral soil substratum of the Haukkasuo bog, where the supply of bases is sufficient to neutralize the acidity.

Conductivity

On most profiles, the conductivities slowly decrease, in parallel with acidity, from the bog surface to the boundary between *Carex* and *Sphagnum* peats (Figs. 9.1, 9.2). Below this boundary, the conductivities increase with mineral content down to the mineral subsoil. The increasing trend of conductivities continues from the *Carex* layer into the clay substratum, indicating the trans-

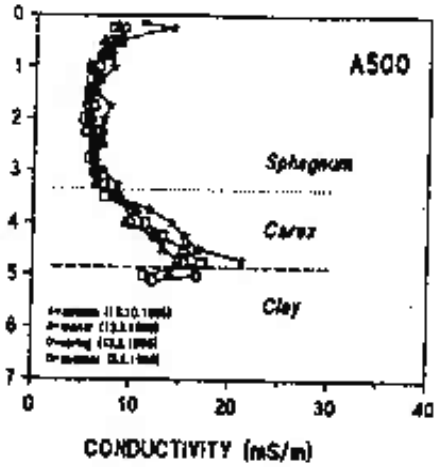

Fig. 9.1 Vertical profile of measured electrical conductivity (mS m⁻¹) by season. Y-axis Depth below peat surface (m)

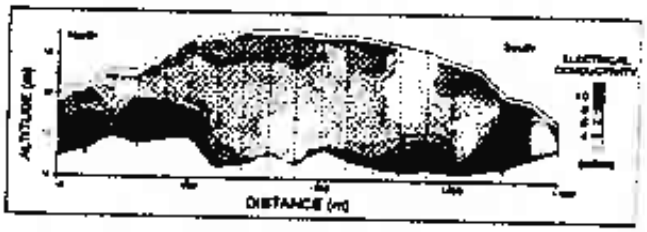

Fig. 9.2 Cross-section through the Haukkasuo bog mire (Finland) with electrical conductivity gradients. Above the *white hatched line*, peat is formed by *Sphagnum* sp., below this line peat is formed by *Carex* sp

ter of ions by diffusion and advection across the peat-clay interface. The conductivity of the basal layers is lower in the central parts of the bog in comparison with the margin, which is clearly seen in the conductivity cross sections of Haukkasuo (Fig. 9.2). The lowest conductivities (about 4 mS m⁻¹), being somewhat higher than the mean conductivity of precipitation in the area (2.3 mS m⁻¹), have been measured in the interior parts of Haukkasuo, extending to the base of the bog in the center. The basal layers of the marginal areas of the Haukkasuo bog are characterized by higher conductivities, rising to the bog surface near its northern and southern margins (Fig. 9.2). In the basal *Carex* peat layers, the conductivity of peat pore water and the pH values are clearly lower near the bog center as compared to the intervening and marginal areas.

The increase of conductivity towards the base is weak near the bog center, where infiltrated surface water flows downwards, but is strong near the margins, where solute-rich groundwater is probably upwelling from the sediments below the peat.

Water flowing in the lower parts of the bog seems to dissolve mineral material and becomes more and more mineralized on the way to the discharge sites at the bog margins. The conductivity pattern proves that a consistent water flow system exists within the bog mire. The measured peat densities of Haukkasuo (50–100 kg m⁻³) correspond approximately to a hydraulic conductivity range of 5×10^{-6} m s⁻¹ to 3×10^{-5} m s⁻¹, according to the relations presented by Päivänen (1973) for Finnish peat. This means that water flux through the lower peat layers in the bog is small and that most of the precipitation and snow melt water discharges through the acrotelm and the loose top peat layers. This confirms the generally accepted concept of slow water movement through bogs.

9.3.3 Modeling a Western Siberian Bog

A three-dimensional (3D) dynamic model for forecasting Holocene and twenty-first century peat and carbon accumulation has been developed for the Bakchar Bog complex, situated in the Southern Taiga zone of Western Siberia. This bog complex consists of various mire types and covers an area of about 800 km², located at the watershed area between the rivers Bakchar and Ilaa (56°49' N, 82°48' E), which are tributaries of the river Ob. The model is based on the (1D) concepts of Ivanov (1981) that: (1) water level doming depends on conductivity and distance to drainage systems (e.g. rivers), (2) peat accumulation follows this doming and (3) development of new peat layers changes the hydrological transmissivity, promoting higher water levels and new peat accumulation, ultimately until water level doming enters equilibrium with discharge capacity. For the 3D approach, a lower conductivity of deeper and older peat layers was supposed. As a result of these concepts, the computed water flow is dominated by surface and sub-surface water flow, according to the findings described above (Minkin 1997).

The model was built using the environmental modeling language PCRaster (Wesseling et al. 1996) and the ground-water modeling code MODFLOW-2000 (Harbaugh et al. 2000). The used time step was 100 years and the horizontal resolution 100 m. Vertically, the model allowed for several layers, i.e. the mineral subsoil and several peat layers. To estimate superficial flow conditions, a vegetation layer (acrotelm) with high conductivity was added. The peat accumulation process started around 6500 years B.P. (Borren et al. 2004).

Besides mineral soil, the bog complex includes four mire types within the study area: marshy peat, through-flow fen, ridge-hollow complex and

ombrotrophic raised bog ('ryam'; see Section 9.2.2). All these five have been implemented in the model.

Note that the peat body is located on a watershed area between rivers, which means that the influence of groundwater and river water is negligible. At places where the water table is above the mineral surface level, marshy peat is formed. As long as the water table is higher than the surface level, the accumulation process continues. As soon as the initial peat deposits are thicker than 0.5 m, a change to other peat types takes place. The transition from one type to another depends on nutrient supply, which in the model was simulated by superficial water-flow conditions. It was assumed that the nutrient supply to vegetation was higher if more water flowed through the vegetation. The main model inputs were hydraulic conductivity, peat growth rate, carbon content, precipitation, evapotranspiration, rivers and digital elevation models (DEM) of the mineral subsoil and the present peat surface. The peat surface DEM is the top of the cartelm.

Table 9.3 shows the used peat growth rates, organic matter densities and carbon contents of the mire types (Bleuten and Lapshina 2001; Borren et al. 2004). Holocene precipitation was derived from the record of Klimanov (personal communication; Fig. 9.3). The net recharge was calculated by multiplying the precipitation with an evapotranspiration factor, which differed

Table 9.3 Properties of various mire and soil types. Peat growth rate, dry organic matter density and carbon content (% of dry organic matter), according to Bleuten and Lapshina (2001) and Borren et al. (2004). Horizontal hydraulic conductivity and evapotranspiration factor (% of precipitation) estimated in this study

Mire or soil type	Peat accumulation rate ($\times 10^{-3}$ m year^{-1})	Dry organic matter density ($\times 10^{3}$ g m^{-3})	Carbon content (% of dry organic matter)	Horizontal hydraulic conductivity (m day^{-1})	Evapo-transpiration factor (%)
Marshy peat	0.5	63.75	54.0	0.0871[a]	85[a]
Through-flow fen	0.35	76.68	53.7	0.0871[a]	85[a]
Ridge–hollow complex	0.39	46.67	53.3	0.0871[a]	85[a]
Ryam	0.42	61.53	47.2	0.0871[a]	85[a]
Mineral soil				3.17	61
Vegetation				337	

[a] One value estimated for all peatland types

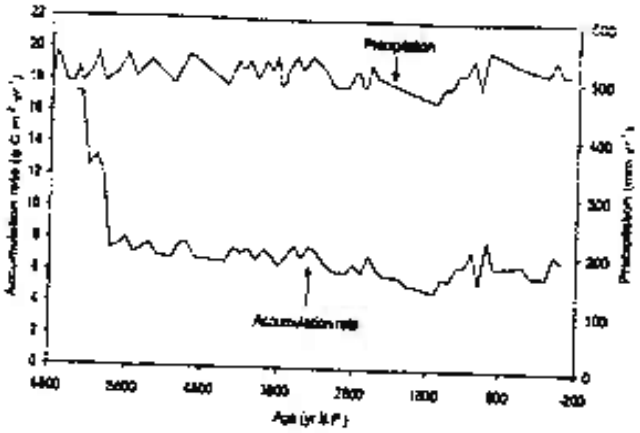

Fig. 9.3 Holocene precipitation record and modeled carbon accumulation; precipitation according to Klimanov (personal communication)

between peat-covered area and mineral soil. The evapotranspiration factors and the hydraulic conductivities of the various layers were estimated by calibrating the model to present hydrological conditions with the PEST algorithm. During a field expedition in August 2002 across the Bakchar Bog, the groundwater level was at the surface (hollows) or 0.1–0.2 m below the peat surface level (ridges, ryam).

Figure 9.4 shows a comparison of the peat distribution over the study area as derived from the DEM, which is the real present situation, and as simulated by the model. At 87 % of the cells, the model prediction, whether there is peat or not, is correct.

The modeled carbon accumulation rate in the study area shows a decrease from 6500 years B.P. to the present (Fig. 9.3). Over this period, a gradual decrease from 17.2 g C m^{-2} year^{-1} to 7.0 g C m^{-2} year^{-1} took place. Obviously, the domed water table between the rivers and streams cannot rise to any extent and will eventually reach a certain maximum (Ingram 1982; Winston 1994). The peat body can retain the incoming water to a limited extent and the lateral expansion will decrease as the peat cover approaches the rivers and streams. The expansion rate is highest at the start of the peat formation, which is shown in Fig. 9.3 by the high carbon accumulation rates.

Fluctuations in the carbon accumulation rate correspond to precipitation as shown by comparison with the precipitation record in Fig. 9.4. The total amount of accumulated carbon in the study area was estimated at 47.4 × 10^{3} g C m^{-2}. This is lower than the mean estimate by Yefremov and Yefremova (2001) of 71.0 × 10^{3} g C m^{-2} for peatlands in the Southern Taiga zone of West-

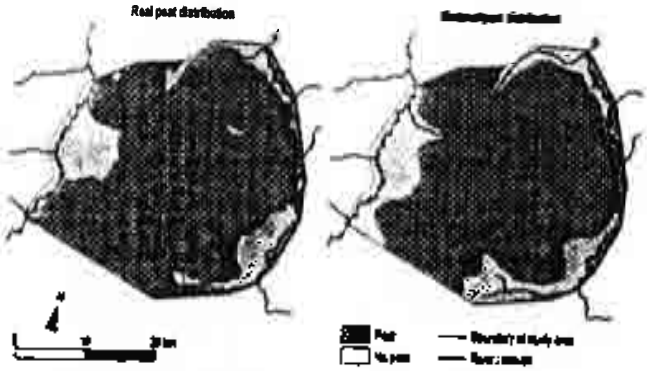

Fig. 9.4 Real and modeled peat distribution within the Bakchar Bog study area. The real situation is based on digital elevation models of mineral subsoil and peat surface

ern Siberia. This difference can be due to the late start of peat formation in our study area. Extrapolation of the model over the twenty-first century gives a near-future carbon accumulation rate of 7.0 g C m^{-2} $year^{-1}$. For the whole study area, this means a carbon accumulation of 4.4×10^9 g C $year^{-1}$.

By this forward modeling, starting from bar mineral soil, the peat thickness and peat presence are predicted quite well, which proves the model concept.

9.4 Fens: Analysis of a Large Pristine Fen in the River Ob Valley

9.4.1 General

Within the vast peatland of Western Siberia, chains (hundreds of kilometers long) of untouched minerotrophic fens are present in the Ob river valley on top of wide Pleistocene terraces which limit the Ob river floodplain. The elevation difference between terraces is 5–20 m. The fens at higher terraces are partly drained for agricultural land use. A section (3–5 km wide, surface area 13 km^2; 56° N, 84° E) belonging to a fen-terrace of ca. 50 km^2 developed on the lower Ob river terraces west from Tomsk was selected to study the hydrology and nutrient conditions (Fig. 9.5).

Peat thickness varies from 5.5 m, in a former river channel eroded in the terrace, near the terrace scarp, to 0.5 m near a secondary Ob channel. By peat accumulation, most of the surface elevation rises above highest Ob river flood

Fig. 9.5 Aerial photograph of the Ob River study site. *Upper left*, upland with forest patches (*dark gray*) and arable land (*light gray*). *Right*, the Ob River. Between the upland and the river is the studied fen on the lower Ob River terrace. The dark patch is a cloud. Numbers indicate the locations of piezometers. Note the darker gray peat ridges stretched half-way up the terrace

level. Climate is severe continental, with 5.5 months of growing season (average summer temperature, June–August: 18 °C) and precipitation in excess of 0.21 m $year^{-1}$, of which 60 % as snow (Bleuten and Lapshina 2001).

Because of the extremely hostile conditions of dense bushes, high water table and weak peat soil, the field study had to be limited to a 3-km transect across the fen from the mineral upland (higher terrace edge) to the nearest river Ob tributary (Fig. 9.5). Along the transect, vegetation relevées have been made and samples taken for estimating the biomass productivity for nutrient content. In piezometer tubes, after coring for peat layer measurements placed in sets of three at different depths, the water level (heads) have been measured weekly from May to September 2002. Transect surface elevation was measured with an optical leveler. Water samples were taken at 14-day intervals and analyzed for main ion components, metals, acidity, alkalinity and conductivity.

9.4.2 Vegetation, Nutrients and Productivity

A close relation between topography, hydrology, hydrochemistry and vegetation has been observed in valley mires where distinct vegetation zones are present parallel to the valley axis (Pulczyński 1984; Heathwaite 1993; Ratcliffe 1977). The width and characteristics of such vegetation zones depend on site conditions, such as the ratio between the supplies of groundwater, surface water and precipitation (Wassen and Joosten 1996; Wassen et al. 2002). In the studied valley fen, a clear vegetation zonation parallel to the valley axis was present. Eight vegetation communities were distinguished, according to their floristic composition (Table 9.4). Based on vegetation characteristics, the studied transect (Fig. 9.6) can be divided into three main zones. Close to the valley edge, a minerotrophic reed zone is located, showing both high-productive, species-poor vegetation dominated by *Phragmites australis* (community 1) and low-productive, species-rich vegetation dominated by *Betula pubescens* (community 2). The open sedge fen located further from the scarp comprises sampling locations 2, 3, 4 and 5 and contains herbaceous vegetation, mainly consisting of sedges (*e.g. Carex lasiocarpa, Carex rostrata*) and *Menyanthes trifoliata*. The mixed sedge–shrub and woodland fen (sampling locations 6, 7, 8, 9) is characterized by the dominance of sedge tussocks (*Carex cespitosa*), trees (*Pinus sylvestris, Betula pubescens*) and shrubs (*Salix rosmarinifolia*) and shows a relatively high total plant species richness. Community 2 combines a high number of species with a low productivity. Communities 3, 4 and 5 show predominantly phosphorus limitation. The peat chemical data show that soil pH is slightly higher near the valley edge (communities 1, 2) and in the open sedge communities (communities 3, 4, 5) than in the forested communities (communities 6, 7, 8).

9.4.3 Hydrology and Modeling

Water flow in fens is of the 'through-flow' type ('Durchströmungsmoore' in Succow and Joosten 2001), which means that most of the horizontal waterflow is at or close to the surface. The groundwater discharges usually into the fens by vertical flow. This concept has been tested by analyses of hydrological field data and modeling.

Using transect data of the cross-section peat profile together with climate data from a nearby meteorological station and river stage data, a 2D steady-state groundwater model was developed based on the MODFLOW code, later enhanced into a 3D model for the entire 15 km² fen area. The 2D model was calibrated with measured groundwater heads and the 3D version with a digital elevation model based on available map data. The measured heads were very close to surface elevation and the dynamics of head variation were very

Table 9.4 Species composition and characteristics of vegetation types (including peat chemical data and hydrochemical data per vegetation type) along the Ob River terrace fen transect. For vegetation data, mean values with standard deviation are given (numbers within brackets indicate number of samples). Mean total numbers of species and mean number of vascular plants are based on relevés made in July 2002 in homogeneous areas of 25 m² (Dierssen 1996). Covers of vascular plants and mosses were estimated using a modified Braun–Blanquet scale (Willmanns 1998). Biomass production was measured by calculating the dry weight of the total standing above-ground living biomass of vascular plants (harvested at 4 July and 9–12 July) after drying at 70 °C for 48 h. After extraction of N, P and K content of vegetation using a Kjeldahl digestion, nutrient limitation was determined by calculating the N:P, N:K and K:P ratios in vascular plants using critical ratios (Olde Venterink et al. 2003). The pH of peat (5–15 cm depth) was determined after Rowell (1997) in rewetted samples in 0.01 M $CaCl_2$ using WTW-pH96. The PO_4 of surface water was determined using ICP

	Vegetation type									
	1	2	3		4		5		6	
Sampling location	0	1	2	3	4	5	6	7	8	9
Distance from terrace (m)	0	80	680	860	960	1180	1650	2030	2500	2850
Vegetation data										
Mean total number of species	18	30	22		16	16	32	40		23
Mean number of vascular plants	16	21	12		12	11	20	21		12
Mean tree cover (%)	0	10	1		0.5	0	30	34		13
Number of relevés	5	14	18		16	10	10	18		14
Biomass production (g m⁻²)	718±180 (5)	322±79 (6)	345±146 (4)		420±71 (4)	530±211 (3)	no data	no data		no data
Nutrient limitation (% P-limited)	33 (3)	0 (6)	75 (4)		50 (4)	67 (3)	no data	no data		no data

Table 9.4 (Continued)

	Vegetation type							
	1	2	3	4	5	6	7	8
Species								
Phragmites australis	V/4	III/2a	-	-	-	-	I/1	-
Angelica archangelica	V/2b	-	-	-	-	-	-	-
Carex lasiocarpa	-	V/2a	V/2a	V/2a	III/2a	I/+	III/1	III/1
Carex rostrata	II/+	IV/1	IV/2a	V/2b	V/2a	V/2b	II/+	-
Menyanthes trifoliata	-	V/3	V/2a	V/3	IV/2b	V/2a	V/2b	-
Betula pubescens	-	V/2a	III/1	II/+	I/+	IV/2a	V/2a	V/2a
Potentilla fruticosa	-	III/+	V/2b	III/1	III/2a	V/2a	IV/1	-
Drepanocladus adunceus	I/+	II/1	IV/2a	III/1	V/3a	II/1	II/1	II/+
Parnassia palustris	-	IV/+	IV/+	III/+	II/+	III/+	III/+	-
Typha latifolia	-	-	-	-	V/3	-	-	-
Pinus sylvestris	-	-	-	I/+	-	V/2b	V/2b	III/+
Carex vesicaria	V/2b	I/+	-	-	-	IV/2a	IV/2a	III/2a
Salix rosmarinifolia	-	-	IV/1	-	II/+	IV/2a	IV/2a	V/2b
pH	7.2±0.15	7.2±0.21	7.0±0.15	6.8±0.27	7.0±0.14	6.5±0.23	6.8±0.23	6.7±0.15
Number of samples	5	14	18	18	10	14	18	16
PO_4 of surface water	3.78±4.10 (3)	1.31±0.89 (15)	2.44±1.36 (17)	2.31±1.34 (15)	2.94±3.02 (10)	1.28±0.80 (8)	0.93±0.52 (17)	0.67±0.37 (6)

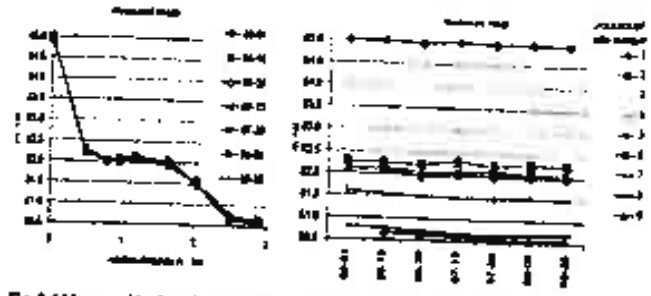

Fig. 9.6 Measured hydraulic heads (groundwater levels) in the Ob River terrace fen from June (06) through August (08). *Left* Along the transect (see Fig. 9.5), *right* time-courses by piezometer

small (0–0.05 m) during the measured period (Fig. 9.6). This means that the 'ground' water level is at or above the surface because 2-week-averaged precipitation was 0.003–0.0047 m day^{-1} and no drains are present. By using the (vertical 2D model) computed groundwater heads as upper boundary conditions, the groundwater flow lines have been calculated (MODPATH). Close to the terrace scarp, a strong groundwater discharge directed to the surface (seepage) takes place near piezometers 1–2 (Fig. 9.7). The seepage water then flows over 'land' in the direction of the Ob river towards piezometer 3, where the water is dammed up by peat ridges, which result in infiltration and downward percolation of surface (over land flow) water. Closer to the Ob, caused by the relative low river stage, infiltration conditions are also present. The measured EC is clearly affected by the seepage water coming from deep and distant upland (Fig. 9.8) The raised EC values could be explained by correlated salt (NaCl) and hydrocarbonate (HCO_3^-) concentrations. The nitrogen (NH_4^--N + NO_3^--N) concentration is generally low (0.1–1.6 g N m^{-3}) in seepage parts of the transect but is raised (2.5–4.0 g N m^{-3}) where infiltration dominates. The low redox conditions in the groundwater of the seepage parts of the cross-section promote denitrification. In the infiltration parts, the relatively oxygenized water nitrates can stay longer in solution.

With the 3D model, water fluxes by over land flow and seepage were calculated and, with PCRaster (Wesseling et al. 1996), were related to the recharge (net precipitation) flux. The outcome of this exercise was the generally low contribution of rainwater to the water balance (0.01–0.3 %) for most of the model area, except on the peat ridges (10–25 %). The contribution of seepage water to the cell water balance was more than 10 % near the terrace scarp and downstream of the peat ridges and, in the rest of the modeled area, lower than 1 %. This means that most water and dissolved nutrients are transported over land between the plants and sedge hummocks.

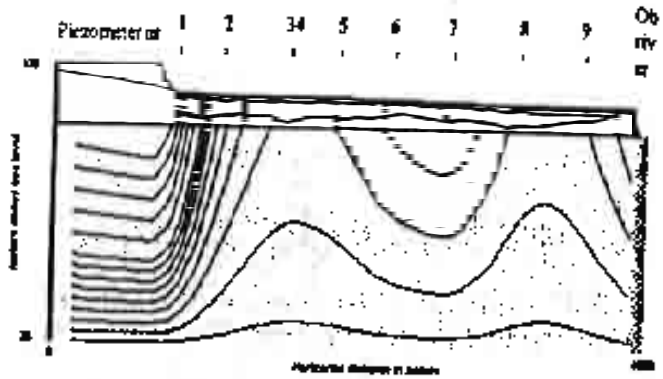

Fig. 9.7 Groundwater flow paths in the vertical cross-section along the measured transect on the Ob River terrace fen

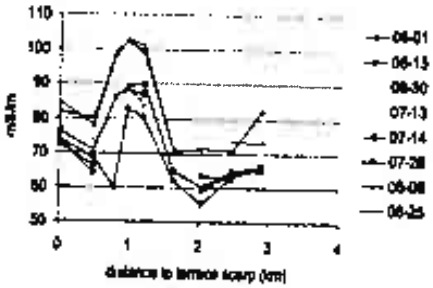

Fig. 9.8 Dynamic and spatial variation in measured peat water electrical conductivity at 1–2 m below the 'surface' along the Ob River terrace fen transect

9.4.4 Hydro-Ecological Integration

Vegetation zonation and hydrochemical characteristics can be explained by hydrological processes. Although there is a strong upward flux of groundwater close to the valley edge, the steep scarp here still is the cause of a significant unsaturated zone, causing relatively dry conditions (Figs. 9.6, 9.7). This may lead to high mineralization rates and more eutrophic conditions. The relatively eutrophic conditions give rise to high-productive, species-poor vegetation dominated by *Phragmites australis* at sampling location 6 and to a high abundance of *Betula pubescens* at sampling location 1. The open sedge fen is also characterized by a strong seepage flux, which in combination with the lower altitude results in relatively high water levels at the mire surface (Fig. 9.6). This

leads to the presence of e.g. sedges and *Menyanthes trifoliata* and to the absen of trees, as they (at least periodically) require water levels below the mire s face (Moore and Bellamy 1973; Wassen et al. 1992). The groundwater seepa flux decreases with increasing distance from the valley edge (Fig. 9.7) and t absence of discharge of regional groundwater in the woodland fen (Fig. 9. prevents constant inundation of the mire surface. This results in water lev near or below the mire surface at sampling locations 6, 7 and 8 (Fig. 9.6) a enables trees to establish in this zone. This analysis shows a number of clo relationships between hydrology, peat chemistry and vegetation zoning. Suc relationships are blurred in peatlands dominated by human intervention an thus can only be studied in relatively undisturbed mires.

9.5 Discussion and Conclusion

In Europe and beyond, the wish to restore bogs and fens is growing because o the high biodiversity values; but because of the slow adaptation of vegetatio to changed hydrological conditions (e.g. draining), the level of degradation i difficult to establish.

Therefore, predicting the effectiveness of rewetting aimed at nature restoration is hampered by a lack of knowledge of the pristine hydrological and hydrochemical (nutrient) conditions of fen ecosystems. Based on vegeta tion, a generalized mire type classification, including two bog types and seven fen types, has been developed. Mainly because of permafrost in the Norther Taiga and Forest Tundra zones, not all mire types are present in the ent south–north gradient across the Western Siberian plain. Related to latitudina climate change, nutrient richness slightly decreases to the North. Nutrient differences between mire types are greater than by latitude.

Bog mires develop in Siberia and North America where groundwater stagnates. This occurs on flat water divide areas and where the drainage system has been changed by geotectonic uplift. In the latter conditions, the 'normal' sequence in a downhill direction from ombrotrophic bog mires via transitional mires, including poor fens and rich fens, to minerotrophic fens can be upset. Water flow on bog mires is predominantly of the 'through-flow' type, which means that most of the surplus precipitation water flows over the peat surface between the mosses and herb roots and tussocks. The dominance of surface runoff over deeper water flow paths has been proved, based on intensive measurements of EC, both vertically and across a Finnish bog mire. This principle has been applied successfully in a 3D MODFLOW-based peat accumulation model of an 800 km² large mire complex in the Southern Taiga of Western Siberia.

Also in fen mires, the dominant water flow is of the through-flow type. As hypothesized, close relations between hydrology, hydrochemistry and vegeta-

tion result in a zonation of vegetation in the Ob valley mires. Due to the dominance of massif influx of groundwater at the upland margin, differences in water chemistry are small throughout the study area and the differences in vegetation characteristics between the zones are correspondingly small. The dominance of groundwater prevents a marked influence of precipitation on vegetation characteristics in the Ob valley mires, resulting in vast areas with quite low-productive (300–500 g dry weight m^{-2}) species-rich (20–30 species per relevée) fen vegetation dominated by sedges such as *Carex lasiocarpa* and *C. rostrata*, forbs like *Menyanthes trifoliata* and *Parnassia palustris* and brownmosses in the major part of the study area.

In the studied Ob river valley fen, apparently a layer consisting of water was present between the top peat and the (floating) roots and green parts of the fen plants and mosses. Most of the nutrients are transported through this top-layer. The nutrient balance showed that discharge of groundwater is much more important than precipitation (by orders of magnitude). The slow adaptation of fen vegetation after cutting canals for drainage can be explained by this typical layer, which can take many years to disappear as horizontal water flow capability gradually decreases during the desiccation process. Vegetation change can be expected when water balance fluxes change, in particular when groundwater influx (seeping-up of groundwater) disappears (Wassen et al. 2002).

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10 Ecological Aspects of Microbes and Microbial Communities Inhabiting the Rhizosphere of Wetland Plants

Paul L.E. Bodelier, Peter Frenzel, Harold L. Drake, Thomas Huber, Kirsten Küsel, Charles Lovell, Patrick Megonigal, Barbara Reinhold-Hurek, Brian Sorrell

10.1 Introduction

Wetlands are among the most important ecosystems on earth. Many fossil fuels on which we now depend were formed in ancient wetlands. At the present time, the high productivity and biodiversity of these ecosystems is of great value for life on our planet. The typical biogeochemical features of wetland soils are at the very centre of the productivity of these systems.

The permanent or periodic flooding of soils or sediments has a number of physical, chemical and biological consequences for the soil/sediment environment (e.g. Ponnamperuma 1984). The most important ramification of the water layer on top of flooded soils and sediments is the restricted entry of atmospheric oxygen. The diffusion of oxygen in water is 10 000 times slower than in air. Oxygen is rapidly depleted in the upper layers of inundated soil due to chemical and biological oxidation processes, resulting in a soil profile where the presence of oxygen is limited to the upper millimetres (e.g. Frenzel et al. 1992). Wetland plants obviously developed mechanisms to equip themselves for colonisation and growth in anoxic flooded soils and sediments, as reviewed elsewhere (Armstrong et al. 1994; Blom and Voesenek 1996). Changes in anatomy, morphology and metabolism are of paramount importance for surviving in anoxic root environments. So-called aerenchymatous tissue (see Section 10.2) in shoots and roots facilitates the transport of oxygen from the atmosphere to the roots, thereby supplying the roots with oxygen for respiration. However, roots also lose substantial amounts of oxygen to the surrounding anoxic soil, as demonstrated in numerous studies (cited by Armstrong et al. 2000). By means of this process of radial oxy-

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EXHIBIT 6

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HYDROLOGIC RESTORATION OF A FEN IN ROCKY MOUNTAIN NATIONAL PARK, COLORADO, USA

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Abstract: Big Meadows, a 63-ha fen in Rocky Mountain National Park (RMNP), was ditched for agricultural purposes in the early part of this century. Although one of the ditch closed after the establishment of RMNP in 1915, it continued to intercept shallow flows in the central and southern portions of the fen, causing the ground-water level to decrease and aerobic soil conditions to develop in the mid- to late-summer of most years. In 1990, the ditch was blocked in an attempt to restore the hydrologic regime in the central and southern portions of the fen. Water-level data from three years prior to restoration and four years after restoration show that blocking the ditch successfully restored surface sheet flow, high late-summer water table levels, and anaerobic soil conditions in much of the central and southern portions of the fen. Conditions in these areas are now similar to those in the northern portion of the fen. The long-term data from the site also indicate that summer rainfall has a greater influence on the magnitude of late-summer drying than the year of the winter snowpack. In a post-restoration year with extremely low rainfall in July and August, water levels throughout the fen decreased to levels similar to those observed throughout most of the pre-restoration period. The study suggests that fens and other fens in the southern Rocky Mountains are extremely sensitive to summer precipitation and the hydrologic changes created by even small ditches or water diversions.

Key Words: fen, hydrology, restoration, Colorado, Rocky Mountains

INTRODUCTION

In many portions of the world, peatlands have been used for fuel or horticultural materials by direct harvesting, or they have been converted to agricultural or silvicultural uses by the construction of drainage ditches. These activities have been well-studied in Europe and Asia (UNESCO 1975, Zoladeski 1973, Stewart and Lance 1983, Conaway et al. 1990, Gilitsch et al. 1993, Heathwaite et al. 1993), Canada (Rubec et al. 1988, Doug and Liefers 1989, Hillman 1992), and in a few areas of the U.S., such as Minnesota, Thayer 1955, Reeder 1972, Olson et al. 1981, Glaser 1987, Gorham and Brooks 1990, Brodel 1992, Kairstead 1993). In contrast, there have been almost no detailed studies on the effects of ditches, water diversions, or mining in peatlands in the western United States or other semi-humid regions.

Peatlands occur where soil saturation retards the decomposition of organic matter, allowing it to accumulate (Moore and Bellamy 1974, Sanger et al. 1996). In humid maritime and some high mountain regions, precipitation alone is sufficient to maintain saturated soils. However, in regions where potential evapotranspiration exceeds precipitation, additional surface or ground-water inflows, or both, are necessary to maintain saturated soils (Cooper 1990, Mitsch and Gosselink 1993, Cooper and Andrew 1994).

The southern Rocky Mountains region has a continental climate with warm, dry summers, and peatlands occur only in areas with consistently high water tables (Cooper 1990). These water tables are sustained by local hillslope drainage (Cooper and Andrew 1994) or local or regional ground-water discharge (Cooper 1996). The dependence of Rocky Mountain peatlands on ground-water discharge means that they are all fens and are often extremely sensitive to climate variability or changes in the supply of ground water due to water diversions (Heathwaite et al. 1993).

Big Meadows is a 63-ha fen in Rocky Mountain National Park that was ditched for agricultural use prior to the establishment of the Park in 1915 (Figure 1). Earlier studies determined that even though agricultural activities ceased when the Park was formed, the ditch was still lowering water tables in the central portion of the fen (Schuster 1988, Cooper 1993). On the

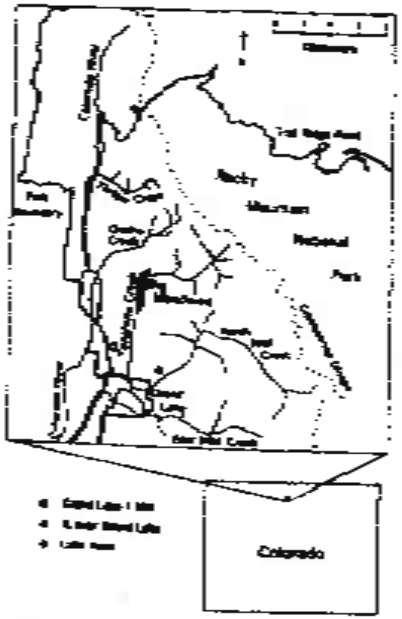

Figure 1. Location of the study area in Rocky Mountain National Park, Colorado.

basis of these investigations, a restoration program was designed and implemented in 1990.

Our hydrologic investigations have continued from 1993 to the present, and in this paper, we analyze three years of pre-restoration and four years of post-restoration water-level data to evaluate both the effects of the ditch and the success of the restoration project. These data provide an important case study of a restoration project in a high elevation peatland and insights into the hydrologic functioning of fen-type fens near the southern limit of their occurrence in North America.

STUDY AREA

The Big Meadows wetland fens upstream of a glacial end moraine in the Tonahutu Creek drainage in the southwestern corner of Rocky Mountain National Park, Colorado at 2,861 m elevation (Figure 1). Tonahutu Creek drains westward from the Continental Divide and is a headwater tributary of the Colorado River. The northern and western area of Big Meadows is a fen supported by ground-water discharging from the toes of adjacent hillslopes, upward seepage from mineral soil, and seepage from spring-fed streams. In the fen portions of the wetland, up to 2 m of peat has accumulated on alluvial and glacial material. Adjacent to Tonahutu Creek, recent alluvial deposits occur at the surface, and no peat is present.

The fen is gently sloping, and its vegetation is dominated by the sedges *Carex aquatilis* Wahlenberg and *C. utriculata* Boott, with the willow *Salix planifolia* Pursh being abundant at the fen margins where mineral-rich ground water discharges from hillside aquifers (Cooper 1990) (nomenclature follows Weber and Wittmann 1996). Conifer forests dominated by *Pinus engelmannii* (Parry) Engelmann, *Pinus ponderosa* Douglas ssp. *ponderosa* (Engelmann) Critchfield, and *Abies lasiocarpa* (A. Murray) occur on adjacent hillsides. The chemistry of surface water in the fen is typical of that found in granitic watersheds in the Rocky Mountains, being circumneutral to moderately acid, with extremely low concentrations of mineral nutrients (Cooper 1990, Cooper and Andrus 1994).

Big Meadows is subject to a strongly seasonal, snowmelt-driven hydrologic regime. In May and June, snowmelt runoff floods the area with up to 10 cm of slowly flowing water. Since summer precipitation is less than potential evapotranspiration, water tables typically decrease during the summer and are deepest from late August through September. The water table begins to rise in late September or October when evapotranspiration decreases. The duration and timing of the summer water-table drawdown is critical, as high water tables are necessary to maintain anaerobic soil conditions and retard the decomposition of organic matter.

In the early 1900s, a ditch was constructed through the central and southern portions of Big Meadows to enhance hay production for livestock. This ditch drains water from the fen to Tonahutu Creek and is approximately 500 m in length, 0.5 m wide, and up to 1.0 m in depth (Figure 2). Although the ditch has not been maintained since 1915, it continues to capture surface water and erode deeper into the peat. Topographic and botanical evidence indicates that, in the absence of the ditch, surface water would flow south through Big Meadows in a water track in path of concentrated water flow (Wright et al. 1992). This water track still exists in the northeastern portion of Big Meadows. We used water levels in that northern area (the area around wells 46 to 60 in Figure 2) as controls to evaluate the effects of climatic differences between the pre- and post-restoration study periods.

The diversion of this surface water, combined with

Figure 2. Location of the ditch in the Big Meadows study area. Also shown are the locations of study wells and Tonahutu Creek. Areas outside the dark line and some wells 38 and 59 are forested.

the lowering of the water table by the incised ditch, extend ground-water levels in the central portion of Big Meadows to drop to more than 40 cm below the soil surface from mid-July through the fall of most years prior to the restoration. The low water table allowed soils to dry and oxidizing conditions to develop. The concern was that oxidizing conditions would increase decomposition rates and result in a long-term loss of peat and substantial ecological changes in the fen.

Since Big Meadows is located 3.5 km from the nearest road, filling the ditch was impossible and was not practical. Instead, 44 pieces of 150-cm-long, 90-cm-wide, and 3-mm-thick galvanized sheet metal were installed across the ditch to block the flow of water in 1990.

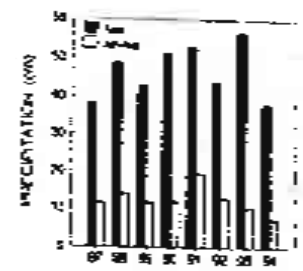

Figure 3. Total water year and June through August precipitation, 1987–1994, recorded at U.S. Weather Service station Grand Lake 2NW. Mean annual precipitation is 39.5 cm, and mean June–August precipitation is 15.0 cm.

METHODS

Hydrology and Climate

Fifty ground-water monitoring wells were installed in Big Meadows (wells 10–60 in Figure 2) in June, 1987. All wells were installed by hand to a depth of 2–2.5 m, and each well was cased with 5-cm-diameter PVC pipe slotted in the native section below the ground surface. Wells were back-filled with native soil and a cap of native clayey soils. Wells were developed

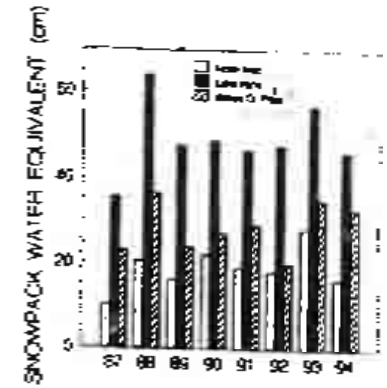

Figure 4. Total snowpack water equivalent for 1 May 1987–1994 for North Inlet, Lake Irene, and Willow Creek Peat stations in Rocky Mountain National Park.

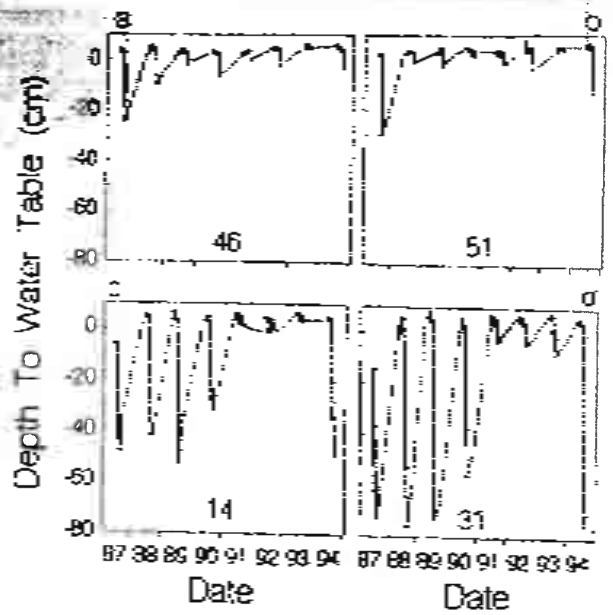

Figure 5. Depth to water table during 1987–1994. Monitoring wells 46 and 51 represent the undisturbed water track, and wells 14 and 31 represent the ditched water track, in Big Meadows. Circles are data points connected by solid lines. Dotted line approximates water-table recovery.

by pumping three cutting reservoirs of water. All wells were monitored approximately weekly from the time Big Meadows became snowfree in early summer through August and less frequently during the fall. Here, we analyze well data from 1 June to 31 August for each year from 1987 through 1989 and 1991 through 1994. Water-level data from 1990 are not analyzed here because they represent the year when the ditch restoration was taking place and therefore don't represent either pre- or post-restoration conditions.

Precipitation and snowpack records were used to determine which pairs of years prior to and following restoration could provide the most valid comparisons. The nearest daily precipitation data were from the U.S. Weather Service station at Grand Lake 1NW (Figure 1), located five km southwest of Big Meadows at 2,780 m elevation. Records at this station date back to 1950. Snowpack water equivalent (SWE) data were available from three stations in the upper Colorado River basin within Rocky Mountain National Park: (1) Lake Irene, located 20 km north of Big Meadows at 3,260 m elevation; (2) Willow Creek Pass, located 15 km west of Big Meadows at 2,907 m elevation; and (3) North Inlet, located 5 km south and slightly east of Big Meadows at 2,740 m elevation (Figure 1). The most comparable pairs of years were 1990/1989, and 1986/1992. These pairs have similar spring SWE values as well as similar amounts and timing of summer precipitation.

Oxidation-reduction potential was measured weekly during June–August 1988–1989 at ten sites in the study area using bright platinum electrodes constructed by welding platinum wire onto copper as described by Faulkner et al. (1989). A total of nine electrodes were installed at each site, with three replicates at 7, 15, and 30 cm depths. Redox values are reported in field measurements corrected for the Calomel reference electrode (DeLaune et al. 1983, Faulkner et al. 1989, Hirsch and Ozanich 1993), not SHE units. Redox potentials greater than +350 mv are considered to indicate oxidizing conditions, while lower values indicate reducing conditions (Sikora and Keeney 1983).

Data Analysis

Depth to water table data for study years were analyzed using agglomerative cluster analysis to determine which wells had the most similar hydrographs for each year. Cluster analysis is an explicit way of identifying groups in your data and is used to classify sites (Jongman et al. 1995). We used the depth to water table on each sample date for each well as the input data to classify wells relative to other wells for each year. The cluster analysis was calculated using Euclidean distance and unweighted pair grouping methods (UPGMA) with the computer program Multivariate Statistical Package 2.1 (Kovach 1993). Dendrograms between paired pre- and post-restoration years were computed to determine which wells shifted position following restoration.

For all study years other than 1990, we determined the number of days between 1 June and 31 August that the water table in each well was above the ground surface and at depths of 0–20 cm, 20–40 cm, and greater than 40 cm, respectively. Field observations of sedge roots indicate that these four categories represent standing water, the upper part of the rooting zone, the lower part of the root zone, and below the root zone, respectively. Tables were made for each year for each depth class using (1) all wells, (2) wells in the ditched portion of the water track, and (3) control wells. The proportion of days between 1 June and 31 August that all wells, water-track wells, and control wells had a water table within each of the four categories described above was compared in pairwise fashion between years using Z-tests (Zar 1984). These analyses were used to indicate whether the restoration project resulted in higher water tables during the summer and the significance of year-to-year differences in precipitation before and after restoration.

RESULTS

Precipitation During the Study Period

Mean annual precipitation for Grand Lake 1NW is 51.3 cm, with a standard deviation of 9.8 cm. Annual precipitation has ranged from 34.8 cm in 1966 to 88.9 cm in 1984. Mean summer (1 June through 31 August) precipitation is 15.0 cm, with a standard deviation of 5.7 cm. Total summer precipitation has varied from 4.4 cm in 1978 to a high of 38.4 cm in 1984. Mean precipitation during June, July, and August is 4.2, 5.3, and 3.5 cm, respectively, with standard deviations of 3.0, 2.5, and 2.8 cm, respectively. These values indicate that summer precipitation is considerably more variable than annual precipitation, and there is more variability for individual months than for the summer as a whole.

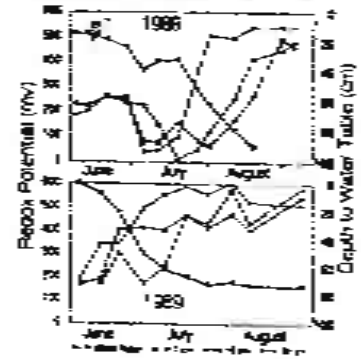

Figure 6. Redox potential at 7, 15, and 30 cm soil depths (means of n = 3 at each depth), and depth to the ground-water table during June–August 1988 and 1989 near well 31.

The years 1987 and 1994 were two of the first drought years on record, with total annual precipitations of 37.6 and 38.0 cm, respectively (Figure 3). The highest total precipitation, 157.6 cm) for the study period was in 1993, but in contrast to 1987 and 1994, this was within one standard deviation of the 45-year mean. The lowest July precipitation total on record (0.9 cm) occurred in 1994, and 1991 had the 5th highest (8.6 cm). Since 1993 and 1994 were both post-restoration years, these differences allowed us to evaluate the effects of high and low mid-summer precipitation on water tables in the control and the restoration areas.

The 1 May SWE values for 1987–1994 indicate that all three stations (North Inlet, Lake Irene, and Willow Creek Pass) are correlated ($r>0.77$, $p<0.05$). Lake Irene, the highest elevation site, consistently has much more SWE than North Inlet, the lowest elevation site (Figure 4). Since most of the annual precipitation accumulates as the seasonal snowpack, late April SWE at these three stations is correlated ($p<0.05$) with total annual precipitation at station Grand Lake 1NW. SWE measurements during the 1988 and 1989 water years indicated that Big Meadows had a higher SWE than Willow Creek Pass but less than Lake Irene.

Hydrologic Analysis

The hydrographs for wells 46 and 51 illustrate the undisturbed hydrologic regime of the water track, as

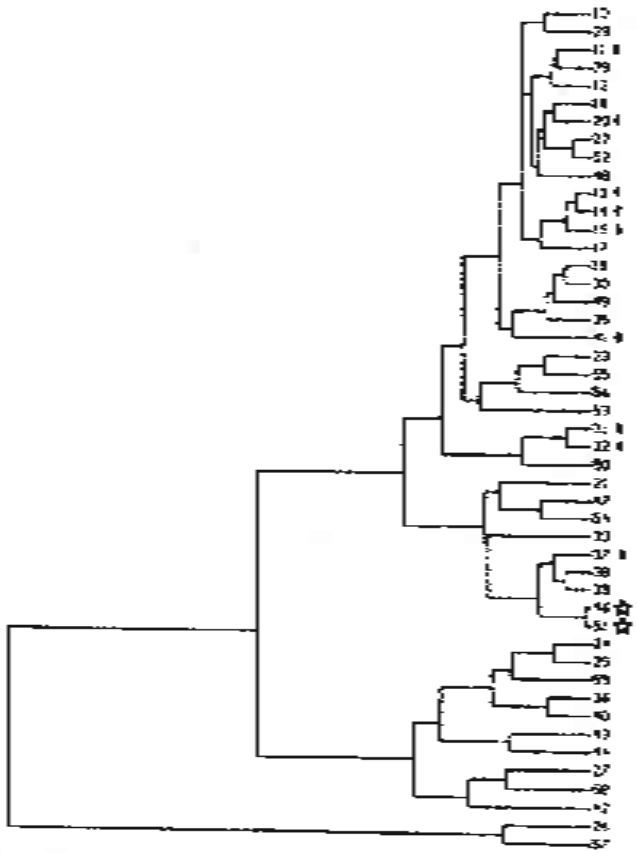

Figure 7. (a) Dendrogram for cluster analysis of soil data for 1988. Wells marked with arrows occur in the ditched water track and do not cluster near control wells 46 and 51 in the undisturbed water track (marked with stars). Maximum Euclidean distance is 487. (b) Dendrogram for cluster analysis of soil data for 1990. Formerly ditched water-track wells cluster together with control wells 46 and 51. Also clustering with these water-track wells are sites outside wells 18, 48, 18, and 28 and well 56 located in a spring. These sites also have water tables near the soil surface for the entire summer in all but the driest years. Maximum Euclidean distance is 502.

the control area of Big Meadows (Figures 5c and 5d). Short flow occurred for most of the summer at these two sites during all eight study years. Only in 1987 and 1994 did the water table drop more than 10 cm below the soil surface for a short period of time in late August. Redox potential measurements at well 46 during 1987–1989 indicate that the soils did not become anoxic (Cooper 1990).

In the ditched water track portion of Big Meadows, the water table at wells 31 and 14 (Figures 5c and 5d)

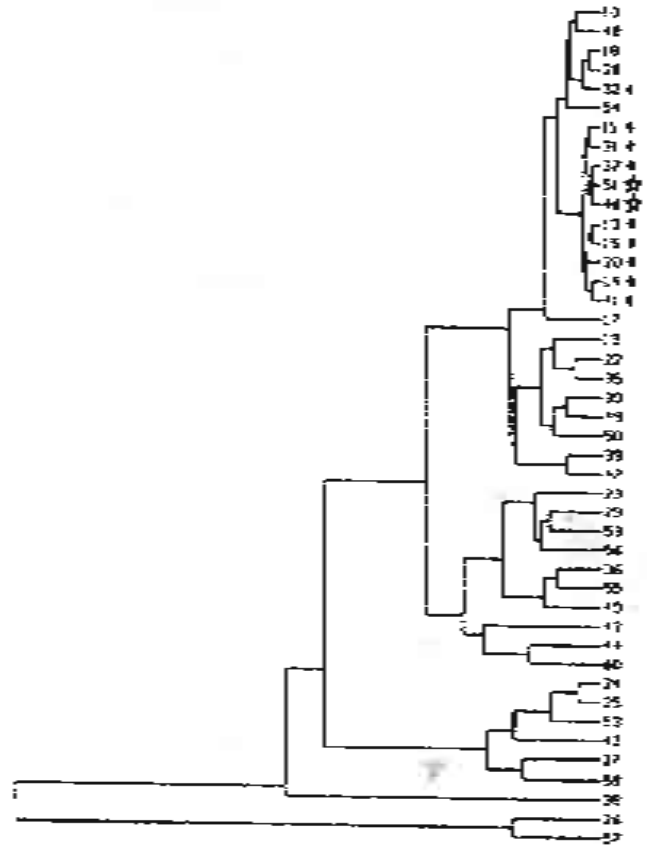

Figure 7. Continued.

dropped to 40–70 cm below the soil surface in July and August in all three pre-restoration years (1987–1989). In contrast, the water table remained near the soil surface for the entire summer during the first three post-restoration years (1990–1992) and was very similar to the water levels in wells 46 and 51. In 1994, the below-normal summer precipitation caused the water table in wells 31 and 14 to drop to levels similar to those observed prior to restoration.

Redox potential measurements taken near well 31 in the ditched portion of the water track during 1988 and 1989 indicated that wetland soils throughout the study area are anaerobic in early June. As the water table dropped, upper soil horizons became aerobic first, while anoxic conditions developed at 30 cm depth 1–2 weeks later (Figure 4). In 1987 and 1988, soils began to oxidize when the water table dropped to more than 35–40 cm below the soil surface. The upper 15 cm of soil became aerobic in late June during the dry year 1989, while during the wet year 1988,

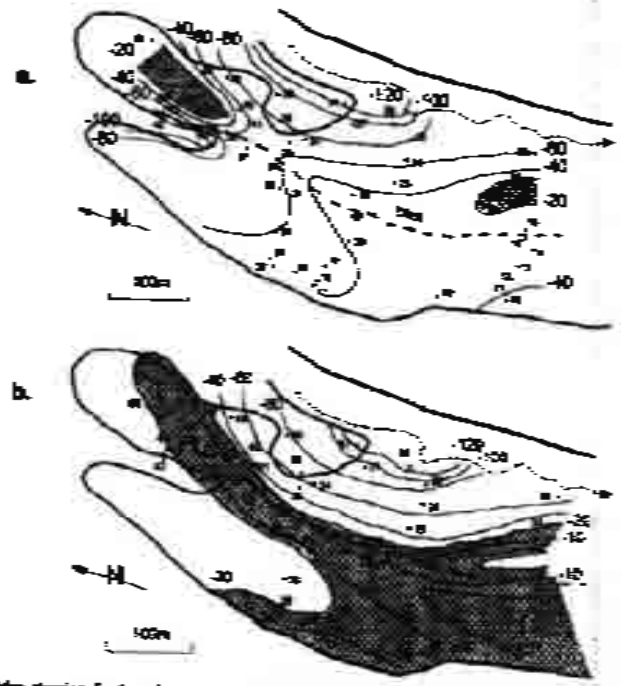

Figure 8. (a) Map showing depth to the ground-water table for 1 August 1989. Shaded areas have a water table within 20 cm of the soil surface. (b) Map showing depth to the ground-water table for 1 August 1991. Shaded areas have a water table within 20 cm of the soil surface and indicate the position of the restored water track.

anoxic conditions did not develop until nearly a month later (Figure 6). Since such as well 46, where the water table rarely drops below 20–30 cm depth, remained anaerobic all summer.

The cluster analysis of well hydrographs for 1989 shows that the wells in the ditched portion of the fen (wells 11, 33, 14, 15, 28, 31, 32, 37, and 43, identified with arrows), are in different portions of the dendrogram from the wells in the control area upstream of the restoration project (wells 46 and 52, marked with stars) (Figure 7a). However, in the 1992 dendrogram, these upper wells were clustered with wells 46 and 52, indicating that their annual hydrographs were very similar (Figure 7b). A similar shift in well positions was observed in a comparison of the 1988 and 1991 dendrograms. Also clustering with the control wells and restored water track wells in the 1992 hydrograph are peripheral margin wells 30, 48, 38, and 13 and one spring site, well 54, all of which have water tables near the soil surface for most of the summer (Figure 7b).

A map of the water table depth below the soil surface in mid-August 1989 shows that the water track in the upstream portion of Big Meadows is truncated by the ditch (Figure 8a). Water tables in the central part of Big Meadows are generally 40–50 cm below the soil surface and would have anoxic soils. In contrast, the depth to water table map for August 1991 (Figure 8b) shows the water table within 20 cm of the soil surface extending through all of Big Meadows except for the portion adjacent to Tenmile Creek, and anoxic soils would occur through this entire area.

Pairwise comparisons between years indicate that water tables in the control wells were at or above the ground surface for a significantly shorter period of time in 1987 and 1989 than 1988 (2-tails, $p<0.05$), 1991, 1992, and 1993 ($p<0.01$), but not than 1994 ($p>0.05$). There was no significant difference in the duration of surface inundation between 1988 and 1993 or between 1989 and 1992, indicating that the comparison of these years using cluster analysis and depth to water table is valid.

When all wells, including the control area and water track, were analyzed together, 1987 and 1989 were the years with the shortest duration of surface inundation and the shortest duration of a water table within 20 cm of the ground surface (Figure 9). There was no significant difference (2-tails, $p>0.05$) in the proportion of days with surface inundation between 1987, 1989, and 1994, yet significant differences did occur between these three years and 1988, 1991, 1992, and 1993 ($p<0.01$).

For wells in the formerly ditched portion of the water track (water-track wells in Figure 9), there was no statistically significant difference in the duration of surface inundation between 1987 and 1989, but these two years were significantly drier than all other years. The wettest pre-restoration year, 1988, was not significantly different from 1994, but 1988 was significantly different than the wettest post-restoration years (1991, 1992, and 1993). Thus, water levels in the ditched water-track area in 1988, a pre-restoration year with above average precipitation, were comparable to the water-table levels observed in 1994, a post-restoration year with below average precipitation.

The comparison of post-restoration water levels between years with average (1992), high (1993), and low (1994) total precipitation illustrates that during average-to-wet years, the water table in the restored water track remains near the soil surface for nearly the entire summer, while in a dry year, such as 1994, the water table dropped to well below the soil surface (Figure 5). Thus, differences in the amount of summer precipitation can result in important differences in wetland pond-water levels. For example, greater than average mid-summer rain in 1991 caused water tables to rise, while in 1994, when little summer rain fell, the water table dropped sharply (Figure 5c) even though winter snowpack was relatively similar between these two years (Figure 4).

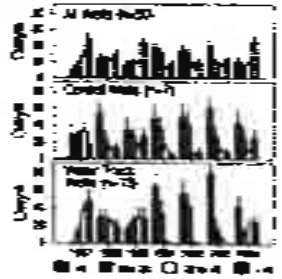

Figure 9. Mean (+1 se.) number of days that all wells, control wells, and wells within the area of the restored water track had a depth to water table above the soil surface (>0), between the soil surface and 20 cm depth, between 20 cm and 40 cm depth, and greater than 40 cm depth for all study years.

DISCUSSION

Since the early 1900s, the ditch in Big Meadows has successfully diverted the water track to Tenmile Creek and thereby removed a major source of water from much of the fen. Analyses of well and redox potential data indicate that prior to 1992, the water table in much of the study area dropped sufficiently far below the soil surface to create anoxic conditions in the upper part of the soil for at least half of the summer in all but high summer-precipitation years.

After blocking the ditch, the water table through the central portion of Big Meadows remained near the soil surface for the entire summer during 1991, 1992, and 1993. Although soil redox potential data are not available during this period, the relationship observed between redox potential and ground-water levels in 1987–1989 indicates that anaerobic conditions would have been minimized by the entire summer. Since water tables were below 20 cm for an average of only about a week from 1991 through 1993 in the restored water track area as compared to nearly seven weeks before restoration, the rate of organic matter decomposition has been greatly reduced, and the fen most likely is accumulating organic matter in all but the driest years.

Between 20 June and 1 August 1994, only 1.1 cm of rain fell in the study area. The lack of precipitation and correspondingly high evapotranspiration caused the water table in the restored water track area to drop to levels similar to the pre-restoration year 1988. This similarity indicates that prior to restoration, soils within the ditched portion of the water track became anoxic even during a relatively high precipitation year such as 1988. We believe that the ditch effectively maintained Big Meadows in a state of severe and prolonged drought for much of the 20th century. The reduction in soil saturation allowed Deschampsia cespitosa (L.) P. Beauvois (tufted hairgrass), a native grass common in seasonally dry mineral soils, to invade the fen. This species has the third highest canopy cover and third highest constancy of any species in the Carex aquatilis-Psychrophila fontinalis (Sir Candolle) We-

bar community type, which occupies the central portion of the fen (Cooper 1990).

°C dates from the base of the peat body indicate that the Big Meadows fen originated after the retreat of Pinedale glaciers nearly 12,000 years BP (Cooper 1990). Although there may have been alternating cycles of peat accumulation and decomposition during the Holocene, spring snowmelt and summer precipitation have been sufficiently consistent to allow the accumulation of 2 m of peat in much of Big Meadows. The construction of the ditch and resultant annual drying of the central fen area may have been one of the most significant environmental changes since the origin of the fen, although it does not appear that subsidence of the peat surface occurred.

Blocking the ditch has, at least temporarily, reduced the effectiveness of the ditch fen diverting surface waters and caused the water table to again flow the entire length of Big Meadows. However, some surface water still flows in the ditch, and this has eroded the soil around some of the metal sheets. Once exposed, this erosion is likely to proceed and threaten the effectiveness of the initial restoration effort. The only permanent remedy is to fill the ditch with soil and revegetate the area with the rhizomatous sedges that dominate the fen.

The data collected in this study shows that fens in the southern and central Rocky Mountains are extremely sensitive to the hydrologic changes that even small ditches or other water diversions can create. Once a ditch is created, it is self-perpetuating, and an active restoration effort must be undertaken to remove the ditch or eliminate the drainage of water by the ditch. Fens in this region do not appear to be sustainable under the drought conditions produced by ditches, water diversions, or ground-water pumping, all of which would lower water tables. We expect that many fens may also be extremely sensitive to even small changes in the amount or timing of summer precipitation. Much of the peat accumulated during the Holocene may have occurred during climatic periods when mid- to late-summer precipitation totals were consistently high.

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Rapid Early Development of Circumarctic Peatlands and Atmospheric CH_4 and CO_2 Variations

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An analysis of 1516 radiocarbon dates demonstrates that the development of the current circumarctic peatlands began ~16.5 thousand years ago (ka) and expanded explosively between 12 and 8 ka in concert with high summer insolation and increasing temperatures. Their rapid development contributed to the sustained peak in CH_4 and modest decline of CO_2 during the early Holocene and likely contributed to CH_4 and CO_2 fluctuations during earlier interglacial and interstadial transitions. Given the decreased tempo of peatland initiation in the late Holocene and the transition of many from fens (which generated high levels of CH_4) to ombrotrophic bogs, a neoglacial expansion of northern peatlands cannot explain the increase in atmospheric CH_4 that occurred after 6 ka.

Modern northern peatlands cover about 4 million km² across Eurasia and North America and represent the biggest wetland complex in the world (Fig. 1). Today, these peatlands are thought to store 180 to 455 Pg of sequestered carbon while also releasing 20 to 45 Tg per year of CH_4 into the atmosphere (*1, 2*). The potential contribution of northern peatlands to fluctuations in atmospheric CH_4 and CO_2 over the late glacial and Holocene, and during earlier interglacials, has been a matter of much speculation and debate (*3–8*).

Ice-core records show that CH_4 concentrations rose from ~350 to 650 parts per billion by volume (ppbv) between the last glacial maximum (LGM), which occurred 20 ka (20,000 calendar years before C.E. 1950), and the Bølling-Allerød (BA) warm period (~15 to 13 ka). They then declined by ~200 ppbv during the Younger Dryas (YD) stadial (~13 to 11.5 ka), rose rapidly to levels over 700 ppbv in the early Holocene (11 to 8 ka), and then declined again between 8 and 6 ka (*3*). It has been maintained that because conditions were not favorable for widespread circumarctic peatland formation until after 8 ka, tropical wetlands or marine clathrates were the likely sources for the CH_4 peak that occurred 11 to 8 ka (*4, 9*). On the basis of the assumed late-Holocene development, it has been suggested that northern peatlands played little role in the declining atmospheric CO_2, which has also been observed during the period from 11 to 8 ka (*5*). Others argue that the development of the northern peatland complex contributed substantially to the early-Holocene CH_4 increase and simultaneously decreased atmospheric CO_2 through carbon sequestration in northern soils (*6–8*).

Resolving the debate on the potential role of the northern peatlands in early postglacial CH_4 variations has become critical since the recent analysis of the deuterium and carbon isotopic composition of CH_4 (δD_{CH_4} and $\delta^{13}C_{CH_4}$) from Greenland ice samples, which suggested that the destabilization of marine clathrates is an unlikely explanation for the BA or early-Holocene CH_4 increases (*10, 11*). In view of this evidence, it has been argued that the sustained high levels of CH_4 that developed at the close of the YD in part require a persistent terrestrial source linked to the warming climate at that time (*11*).

Holocene concentrations of atmospheric CH_4 reached a minimum of <600 ppbv at 6 ka and then increased again over the late Holocene to values of about 695 ppbv just before the industrial revolution (*3*). This late-Holocene increase has been variously attributed either to expansion of northern wetlands due to neoglacial climatic cooling after the Holocene thermal maximum (*4*) or to the product of expanding anthropogenic activity (particularly the expansion of rice- and cattle-based agrarian societies) in the mid- to late Holocene (*12*). However, recently collected CH_4 data from Antarctic ice cores reveal that the mid- to late-Holocene increase is not unique. A similar late-interglacial increase in Pleistocene atmospheric CH_4 occurred ~400 ka during Marine Isotope Stage 11 (MIS11), which clearly cannot reflect anthropogenic sources and has been ascribed instead to natural factors, including expansion of northern wetlands (*13*).

To address the hypothesis that northern peatland development could have contributed to the late-Pleistocene and Holocene variations in atmospheric CH_4 and CO_2 outlined above, we collated 1516 basal radiocarbon dates for peat initiation from wetlands throughout high-latitude Europe, Asia, and North America from a wide variety of sources (*14*). Some areas, such as Fennoscandia, have numerous basal dates for a small geographic area, whereas other very large areas such as central and eastern Siberia have a limited number of dates (Fig. 1). Therefore, we analyzed the compiled data set by raw number of initiation dates, and we also divided the Northern Hemisphere into grids of 2° latitude by 2° longitude and assigned a value for peatland initiation based on the oldest basal radiocarbon date in each cell (Fig. 1).

The lack of basal dates older than about 16.5 ka suggests that there was no extensive peatland complex in the northern circumpolar region during the LGM (Fig. 2). This finding is corroborated by palynological data that indicate a paucity of *Sphagnum* (peat moss) spores from deposits of this age (*15*). Before 16.5 ka, much of the North American and European arctic and subarctic were still covered in ice, and it is likely that the large ice-free areas of Siberia and Beringia were too cold and dry (*16*) to promote extensive peatland development. This absence of any significant northern peatland complex during the LGM is consistent with the depressed CH_4 levels and the relatively low proportion of northern CH_4 sources observed in ice-core records (Fig. 3).

In concert with increasing summer insolation and northern high-latitude temperatures, the current northern peatland complex began developing in ice-free portions of North America and Asia between 16.5 and 14 ka and initiating widely on all three northern continents after 14 ka (Figs. 1 and 2). These results dispel the earlier assertion that peatland development in

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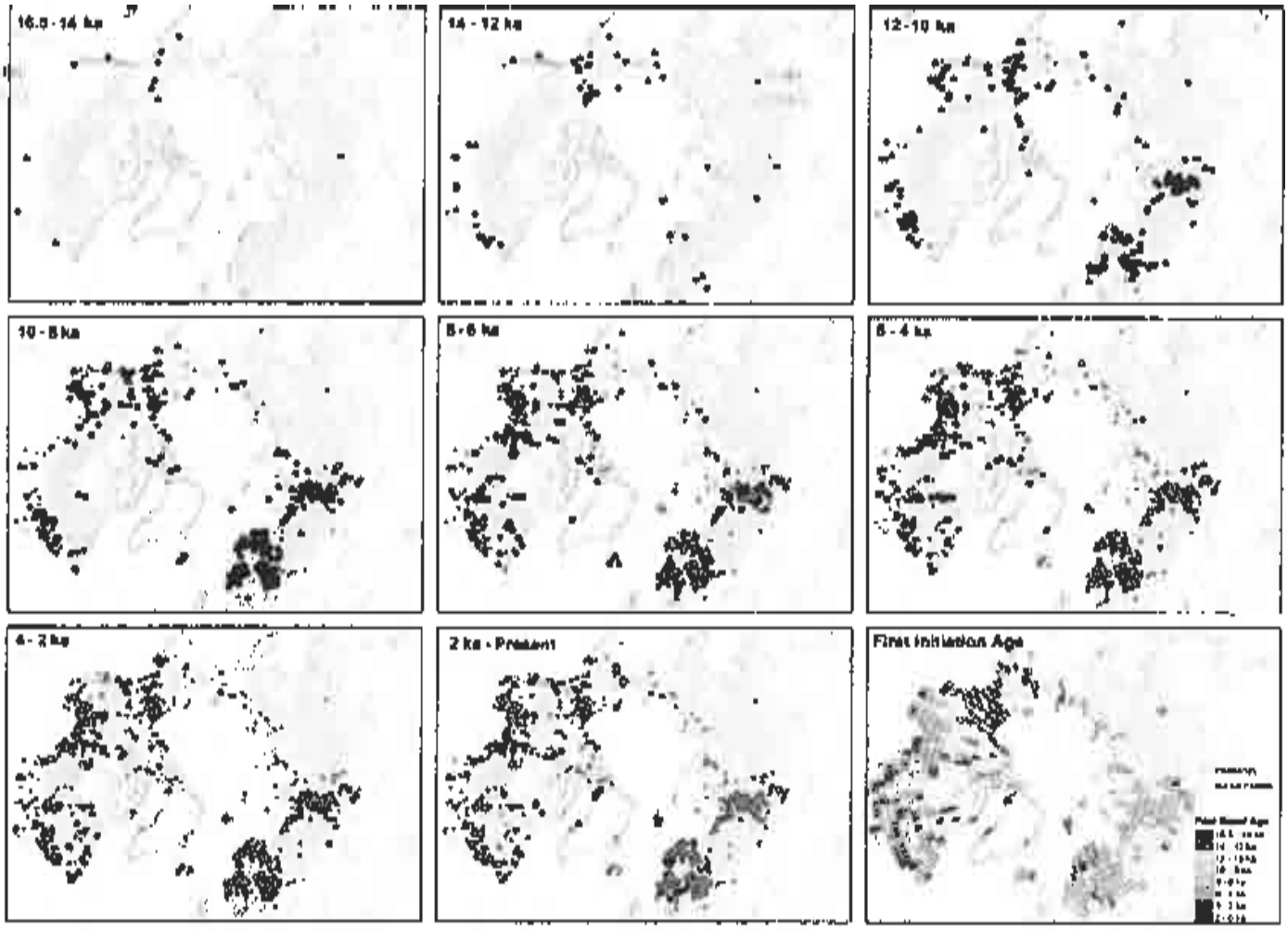

Fig. 1. Map of the current distribution of the northern circumarctic peatland complex and the boreal forest biome developed from a number of sources (SOM text) and the initiation dates of the peatlands based on radiocarbon dates from the base of peat deposits (*14*). The red dots indicate new peatlands that initiated during each time slice and the gray dots indicate preexisting peatlands initiated during earlier time slices. The timing of peatland initiation based on 2° by 2° grids is shown in the right panel in the bottom row. The grids map the oldest basal peat dates within each grid cell.

western Siberia was in advance of development in North America or elsewhere (*7*). The initial expansion coincides with warming during the BA period (Fig. 3), and it is likely that increasing warmth and moisture and decreasing glacial ice cover promoted peatland growth. Resultant development of the northern peatland complex corresponds with increasing atmospheric CH_4 concentrations observed during the BA (Fig. 3). However, the still-limited extent of northern peatlands at this early interval is consistent with evidence for a relatively restricted role of northern wetlands in producing the BA CH_4 increase (Fig. 3). There is a small decline in the rate of new peatland formation during the YD, which may be attributed to the development of cold conditions over much of the Northern Hemisphere and readvances of ice. At the same time, there is a precipitous drop in CH_4 recorded in the ice cores (Fig. 3) that seems to reflect additional factors. During the BA and YD periods, it is unlikely that the extent and growth of the northern peatlands was great enough to have significantly affected atmospheric CO_2 concentrations.

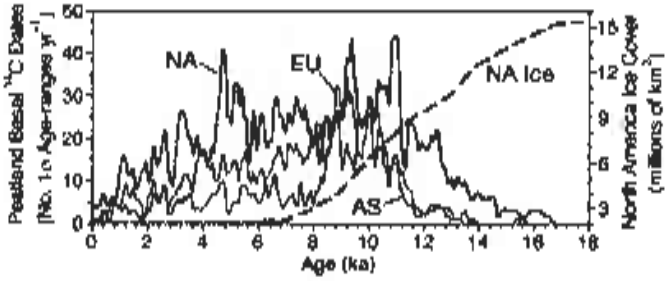

Fig. 2. Timing of circumarctic peatland establishment in North America (black), Europe (red), and Asia (blue) based on the total number of initiation dates from each region. The occurrence frequency of basal peat radiocarbon ages is plotted as the number of calibrated age ranges that fall in any year (*14*), smoothed with a 100-year running mean. The decreasing area of the Laurentide Ice Sheet (dashed line) as it retreated over the late glacial and Holocene (*17*) is also plotted.

The initiation of the early Holocene warming at 11.5 ka after the YD is marked by rapid expansion of peatlands throughout the north. The North American peatlands expanded rapidly during this time (Figs. 1 and 2), even though still constrained by the ice extent of the Laurentide Ice Sheet (*17*). The tempo of subsequent northern peatland development in North America was influenced by the rate of continued ice retreat and the exposure of land surfaces (Fig. 2). Large areas of northern Asia never supported ice during the LGM (*18*), and by 11 ka, high-latitude climate had warmed (Fig. 3). What remained of the Scandinavian Ice Sheet was restricted to cen-

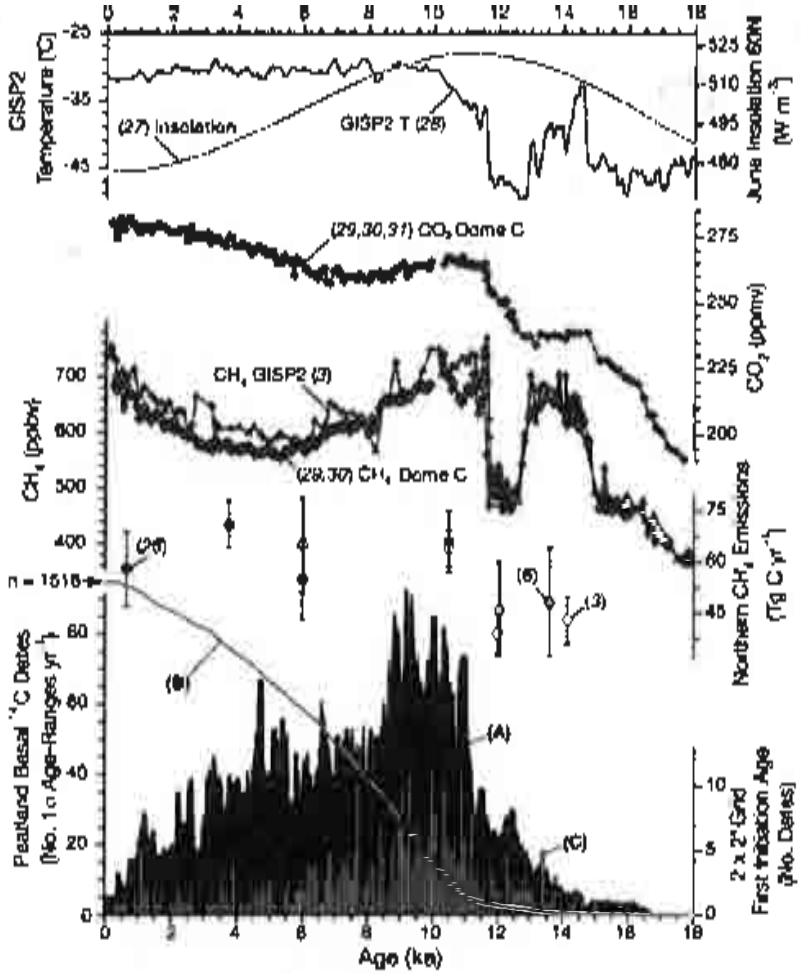

Fig. 3. Timing of circumarctic peatland establishment compared with June insolation at 60°N (*26*), Greenland Ice Sheet Project 2 (GISP2) temperature reconstruction (*27*), atmospheric CO_2 and CH_4 concentrations, and estimates of Northern Hemisphere CH_4 emissions derived from the InterPolar CH_4 Gradient (IPG) (*3, 4, 6, 28*). Atmospheric CO_2 and CH_4 concentrations show ice-core data from European Project for Ice Coring in Antarctica (EPICA) Dome C (red for CO_2 and blue for CH_4) and GISP2 [green triangles show CH_4 (*3, 29–31*)]. Dome C data are shown on the original EPICA Dome C 1 time scale from 0 to 10 ka before present (circles) and on the GISP2 time scale with CH_4 synchronization from 10 to 18 ka before present (diamonds; SOM text). Dome C error bars indicate 1σ uncertainty. (**A**) The occurrence frequency of 1516 radiocarbon dates of basal peat deposits (*14*) shows the number of calibrated age ranges that occur in any year (black curve). (**B**) Cumulative curve of 1516 dates (red curve). (**C**) The oldest basal peat dates within each 2° by 2° grid (gray bars).

tral Fennoscandia (*19*), allowing widespread expansion of peatlands in northern Eurasia (Figs. 1 and 2). A sustained period of maximum rates of peatland establishment followed and persisted until about 8 ka.

The rapid expansion of peatlands at the close of the YD coincides with a ~15-Tg increase in atmospheric CH_4 derived from the Northern Hemisphere (Fig. 3) and a 200- to 250-ppbv increase in total CH_4 concentrations. According to our data, the northern peatland complex was likely at ~20% of its current areal extent at the end of the YD and expanded to about 50% by 8 ka. On the basis of current estimates of overall CH_4 production from northern peatlands (*1, 2*), they may have contributed 4 to 9 Tg of Northern Hemisphere CH_4 after the end of the YD and up to 12 to 27 Tg by 8 ka. However, based on typical peatland succession stages, higher summer insolation in the early Holocene, and evidence of peatland vegetation and type from our peat cores taken in Siberia (*7*), it is likely that many of these newly developed peatlands were warm and wet minerotrophic fens, often dominated by sedges. Such fens typically emit CH_4 at rates many times greater than the ombrotrophic *Sphagnum* bogs common in much of the north today (*6, 20–22*). Therefore, we suspect that the rates of CH_4 production in northern peatlands may have been considerably higher in the early Holocene than they are today.

The $\delta^{13}C_{CH_4}$ values of ice from the Pakitsoq outcrop in western Greenland have been used to infer the likely origins of early-Holocene atmospheric CH_4 (*11*). It has been suggested that major terrestrial sources contributing methane during the period from 11 to 8 ka likely produced CH_4 with $\delta^{13}C$ values of between −50 and −60 (*11*). Although northern peatlands dominated by ombrogenous bogs may typically emit CH_4 with $\delta^{13}C$ < −60, pore waters and emissions of boreal fens are relatively enriched in ^{13}C (average $\delta^{13}C_{CH_4}$ values of −50 to −60), particularly in high-productivity sites (*23*). These values are consistent with the early northern peatland complex, dominated by minerotrophic fens, which was a major contributor to the peak in CH_4 that occurred 11 to 8 ka and must be considered in addition to the potential tropical sources suggested previously (*11*).

The rapid growth of the circumarctic peatland complex and its associated sequestration of phytomass carbon in the early Holocene may have also contributed to the decline by ~7 parts per million by volume (ppmv) of atmospheric CO_2 observed in the ice-core records between 11 ka and the mid-Holocene (Fig. 3). This decline has been interpreted as reflecting a total biosphere uptake of 110 Pg C in the first half of the Holocene (*8*). More than half of the peatland basal dates in our data set are older than 8 ka, indicating rapid initiation and development of this carbon sink in the early Holocene. Conservatively, if by 8 ka peat deposits were 0.5 to 1 m thick, covered just one-quarter of today's peatland area, and were similar in carbon characteristics to today's northern peatlands (*1*), they would have been capable of sequestering 29 to 58 Pg C. Further detailed reconstruction of net peat accumulation rates during this period are required to better estimate the magnitude of the contribution of the northern peatlands to early-Holocene atmospheric CO_2 declines.

The widespread development of peatlands in response to increasing summer insolation, BA warming, and particularly rapid Holocene warming after the YD supports the hypothesis that the northern peatlands were a major terrestrial factor contributing to the early fluctuations in atmospheric CH_4 and to CO_2 sequestration during the current interglacial. It is likely that they played a similar role in earlier interglacials. The rapidity and large spatial extent of the response of the northern peatland complex, particularly to the onset of post-YD warming, suggests that they may even have played a role in CH_4 and CO_2 variations at the shorter time scale of earlier interstadials such as MIS3.

The observed decline in peatland initiation, particularly in Europe and Asia after 8 ka, corresponds to a decline in atmospheric CH_4 concentrations between 8 and 6 ka (Fig. 3). This decline in initiation rates should not be confused with a decline in total peatland area; most peatlands that were extant at 8 ka were still extant at 6 ka. However, many had trans

formed from early minerotrophic fens to ombrotrophic *Sphagnum* bogs, which are typically weaker sources of CH_4 than are fens (*20–23*). Therefore, the transformation from high CH_4-efflux fens to *Sphagnum* bogs, coupled with a declining rate of new peatland formation, would have contributed to the decline in atmospheric CH_4. What remains puzzling is the role that northern peatlands played in the subsequent increase in CH_4 between 6 ka and just before the Industrial Revolution. By C.E. 1700, levels of atmospheric CH_4 had increased once again to almost 700 ppbv. However, contrary to earlier speculation (*9, 15*), new peatland initiation was relatively modest in the late Holocene, and conversion of fens producing high levels of CH_4 to *Sphagnum* bogs with lower production was ongoing. If the mid- to late-Holocene CH_4 increase does not have an anthropogenic explanation, then its source must lie in factors other than large-scale resurgent expansion of the northern peatland complex.

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Supporting Online Material

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SOM Text
Table S1
References

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Gold in Magmatic Hydrothermal Solutions and the Rapid Formation of a Giant Ore Deposit

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The Ladolam hydrothermal system, on Lihir Island, Papua New Guinea, hosts one of the youngest and largest gold deposits in the world. Several deep (more than 1 kilometer) geothermal wells were drilled beneath the ore bodies to extract water at >275°C and to facilitate open-pit mining. Using a titanium down-hole sampler, we determined that the deep geothermal brine of magmatic origin contains ~15 parts per billion gold. At the current gold flux of 24 kilograms per year, this deposit could have formed within ~55,000 years. The combination of sustained metal flux and efficient metal precipitation led to the formation of a giant hydrothermal gold deposit in a short period.

The origins of giant hydrothermal gold deposits are enigmatic (*1*). This is because the concentrations of precious metals and flow rates of ore-forming fluids are poorly quantified, and the origins of the metals are unclear. These aspects can be clarified with direct analyses of fluids from modern hydrothermal systems. The only known active hydrothermal gold deposit is at Ladolam, on Lihir Island, Papua New Guinea, and this deposit is one of the largest in the world with ~1300 tons of gold. Geothermal drilling at Ladolam has been ongoing since before the start of open-pit mining in 1997, and the wells provide access to the deep fluids upstream of the ore zone. Because precious metals precipitate in the wells during fluid ascent due to boiling (*2*), we used a titanium down-hole sampler to obtain deep fluids for analyses (*3*). Any metals that precipitate in the sampler during its return to the surface can be incorporated into the water sample by rinsing the titanium sampler with strong acid. Thus, the deep metal concentrations can be directly measured and used to constrain the rate at which the gold is transported and then deposited in order to form this giant deposit. Because the isotopic compositions of the deep hydrothermal waters at Ladolam are predominantly magmatic in origin (*4*), our data cover not just the origin of the metals, but also the concentrations of gold and related metals in magmatic hydrothermal solutions, which are thought to be important to the formation of ore deposits (*5, 6*).

The Ladolam gold deposit occupies the center of the extinct Luise volcano on Lihir Island (Fig. 1). The two major tabular ore zones cover ~2 km² and extend from the surface to 400 m below sea level. They lie in the middle of a breached crater that formed in response to sector collapse and unroofing of the volcanic edifice ~400,000 years ago (*4, 7, 8*). The resulting explosive depressurization of the magmatic-hydrothermal system produced a diatreme breccia complex and highly permeable rocks, which now host the ore. Mineralogical, fluid inclusion, and isotope studies (*4*) show that the gold was deposited in two stages between 150° and 250°C, from solutions of magmatic origin, when they mixed with other fluids or boiled. The magmatic gold-bearing solutions were near neutral to slightly alkaline pH and contained 5 to 10 wt % equivalent NaCl. The host rocks comprise alkalic mafic to intermediate volcanic and intrusive rocks (*9*) that have been hydro-

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EXHIBIT 8

Ground-Water Quality in Unmined Areas and Near Reclaimed Surface Coal Mines in the Northern and Central Appalachian Coal Regions, Pennsylvania and West Virginia

By Steven D. McAuley and Mark D. Kozar

National Water-Quality Assessment Program

Scientific Investigations Report 2006-5059

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
DIRK KEMPTHORNE, Secretary

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FOREWORD

The U.S. Geological Survey (USGS) is committed to providing the Nation with credible scientific information that helps to enhance and protect the overall quality of life and that facilitates effective management of water, biological, energy, and mineral resources (http://www.usgs.gov/). Information on the Nation's water resources is critical to assuring long-term availability of water that is safe for drinking and recreation and is suitable for industry, irrigation, and fish and wildlife. Population growth and increasing demands for water make the availability of that water, now measured in terms of quantity and quality, even more essential to the long-term sustainability of our communities and ecosystems.

The USGS implemented the National Water-Quality Assessment (NAWQA) Program in 1991 to support national, regional, State, and local information needs and decisions related to water-quality management and policy (http://water.usgs.gov/nawqa). The NAWQA Program is designed to answer: What is the condition of our Nation's streams and ground water? How are conditions changing over time? How do natural features and human activities affect the quality of streams and ground water, and where are these effects most pronounced? By combining information on water chemistry, physical characteristics, stream habitat, and aquatic life, the NAWQA Program aims to provide science-based insights for current and emerging water issues and priorities. From 1991-2001, the NAWQA Program completed interdisciplinary assessments and established a baseline understanding of water-quality conditions in 51 of the Nation's river basins and aquifers, referred to as Study Units (http://water.usgs.gov/nawqa/studyu.html).

In the second decade of the Program (2001-2012), a major focus is on regional assessments of water quality conditions and trends. These regional assessments are based on major river basins and principal aquifers, which encompass larger regions of the country than the Study Units. Regional assessments extend the findings in the Study Units by filling critical gaps in characterizing the quality of surface water and ground water, and by determining status and trends at sites that have been consistently monitored for more than a decade. In addition, the regional assessments continue to build an understanding of how natural features and human activities affect water quality. Many of the regional assessments employ modeling and other scientific tools, developed on the basis of data collected at individual sites, to help extend knowledge of water quality to unmonitored, yet comparable areas within the regions. The models thereby enhance the value of our existing data and our understanding of the hydrologic system. In addition, the models are useful in evaluating various resource-management scenarios and in predicting how our actions, such as reducing or managing nonpoint and point sources of contamination, land conversion, and altering flow and (or) pumping regimes, are likely to affect water conditions within a region.

Other activities planned during the second decade include continuing national syntheses of information on pesticides, volatile organic compounds (VOCs), nutrients, selected trace elements, and aquatic ecology, and continuing national topical studies on the fate of agricultural chemicals, effects of urbanization on stream ecosystems, bioaccumulation of mercury in stream ecosystems, effects of nutrient enrichment on stream ecosystems, and transport of contaminants to public supply wells.

The USGS aims to disseminate credible, timely, and relevant science information to address practical and effective water-resource management and strategies that protect and restore water quality. We hope this NAWQA publication will provide you with insights and information to meet your needs, and will foster increased citizen awareness and involvement in the protection and restoration of our Nation's waters.

The USGS recognizes that a national assessment by a single program cannot address all water-resource issues of interest. External coordination at all levels is critical for cost-effective management, regulation, and conservation of our Nation's water resources. The NAWQA Program, therefore, depends on advice and information from other agencies—Federal, State, regional, interstate, Tribal, and local—as well as nongovernmental organizations, industry, academia, and other stakeholder groups. Your assistance and suggestions are greatly appreciated.

Robert M. Hirsch
Associate Director for Water

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Conversion Factors and Datum

Multiply	By	To obtain
	Length	
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
square mile (mi^2)	2.590	square kilometer (km^2)
	Flow rate	
million gallons per day (Mgal/d)	0.04381	cubic meters per second (m^3/s)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F = (1.8 \times °C) + 32$$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$°C = (°F - 32) / 1.8$$

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Abbreviated water-quality units: Chemical concentration and temperature are given in metric units. Chemical concentration is expressed in milligrams (1×10^{-3} grams) per liter (mg/L), micrograms (1×10^{-6} grams) per liter (µg/L), or picograms (1×10^{-12} grams) per liter (pg/L)

Ground-Water Quality in Unmined Areas and Near Reclaimed Surface Coal Mines in the Northern and Central Appalachian Coal Regions, Pennsylvania and West Virginia

by Steven D. McAuley and Mark D. Kozar

Abstract

Findings are presented from investigations during 1996-1998 by the U.S. Geological Survey National Water-Quality Assessment Program. Ground-water quality in 58 wells downgradient of reclaimed surface coal mines is compared to ground-water quality from 25 wells in unmined areas (background concentrations) in the bituminous coal fields of the northern Appalachian coal region (high-sulfur coal region) in Pennsylvania, Maryland, and West Virginia and the central Appalachian coal region (low-sulfur coal region) in West Virginia. Ground water in the mined high-sulfur coal region has significantly greater median concentrations of sulfate, hardness, calcium, and specific conductance compared to the unmined high-sulfur coal region and to both mined and unmined areas in the low-sulfur coal region. Ground water in mined areas had median values of mine-drainage constituents (sulfate, iron, manganese, aluminum, hardness, calcium, magnesium, turbidity, and specific conductance) that were significantly greater than medians for wells in unmined areas. Mine-drainage constituents include cations such as calcium and magnesium that become elevated compared to levels in unmined areas because of exposure of acidic mine drainage to calcareous materials. The transport of pyrite-oxidation products from the mined site and subsequent neutralization reactions by calcareous materials at the mine site or along the flow path are likely processes that result in greater concentrations of mine-drainage constituents in mined areas compared to unmined areas. Mine-drainage constituents generally exceeded unmined-area background concentrations within about 500 feet of mined sites but were at or below background levels in wells more than 1,000 feet downgradient of mined sites. Concentrations of sulfate, hardness, and total dissolved solids were greatest at well depths of 50 to 150 feet but generally were less than background concentrations in wells deeper than 150 feet. Concentrations of iron, manganese, and aluminum exceeded background concentrations in many wells less than 150 feet deep.

In mined areas, median ground-water ages are nearly as old in hill locations as in valley locations. Older ground-water age correlates with increased distance from mined areas. The lack of significant correlation among mine-drainage-constituent concentrations, ground-water age, distance from mined areas, and topographic locations may be the result of factors such as (1) mixing of ground-water ages in wells open to fractures with variable depths, lengths, and interconnections; (2) disturbance of rock from blasting; and (3) variations in slope and terrain relief in the study area.

Introduction

The overall goal of the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program is to describe the status and trends in the quality of the ground-water and surface-water resources of the United States and to link these trends to an understanding of the natural and human factors that affect the quality of these resources (Gilliom and others, 1995). The NAWQA Program integrates water-quality information on both a national and a local scale. Major components of the ground-water portion of the program are to 1) assess the quality of ground water in major ground-water regions throughout the United States and 2) determine the factors that affect the quality of water in these aquifer systems. This goal is achieved primarily by conducting surveys to assess the occurrence and distribution of water-quality constituents within an aquifer and to link data collected as part of the surveys to the natural and human factors that affect the quality of water within the aquifer. A study of ground-water quality near surface coal mines provides data on the effects of mines on ground-water quality.

Sulfur, iron, and manganese are sometimes present in high levels in coal, coal overburden, and rock that underlies the coal (Brady, 1998). Coal-mine drainage can be acidic or alkaline. Oxidation of iron-disulfide minerals, such as pyrite (FeS_2), occurs when the coal and overburden are exposed to oxygen and moisture, resulting in increased levels of sulfuric acid, iron, manganese, and aluminum in ground water. Pyrite oxidation is the principal mechanism for the formation of acidic mine drainage (AMD), in which acidity exceeds alkalinity. Neutral or alkaline mine drainage has alkalinity that equals or exceeds acidity but can still have elevated concentrations of sulfate, iron, and manganese. Because sulfate is conservative in chemi-

cal reactions and not normally targeted for removal by treatment, it is generally expected to persevere in mined areas (Rose and Cravotta, 1998).

Surface coal mines (surface mines) have had major effects on the quality of shallow ground water, especially with respect to iron, manganese, sulfate, and turbidity levels (Rauch, 1987; Williams and McElroy, 1995). The effects of blasting during surface mining also can have negative effects on ground-water supplies by causing turbidity levels in wells to increase and by causing a loss of water to wells and springs (McCurry and Rauch, 1986). Other problems commonly reported as a result of surface mining include changes in ground-water quality due to increased levels of methane and (or) hydrogen sulfide gas. These problems are significant because the vast majority of rural homeowners in the study area obtain their water from ground-water sources. In some instances, the effects of mining on ground water have caused rural homeowners to abandon their wells or springs, to purchase bottled water, or to haul water from other locations.

In 1977, Congress passed the Surface Mining Control and Reclamation Act (SMCRA). Some provisions of SMCRA include (1) regrading benches formed during the mining process to approximate original contour of the land prior to mining; (2) installing sedimentation ponds prior to and during mining to reduce sediment loads to receiving streams; (3) revegetating reclaimed strip benches upon completion of mining; (4) stabilizing and protecting surfaced-mined areas including spoil piles to control erosion and reduce the potential for surface and ground-water contamination; and (5) managing debris, toxic or acid-forming materials by burying, compacting, or disposing of the material in such a manner so as to prevent ground- or surface-water contamination (U.S. Office of Surface Mining Reclamation and Enforcement, 1993). In concept, changes in mining practice as a result of SMCRA should result in an improvement in surface- and ground-water quality.

In the Pennsylvania Code, Title 25, Subsection 87.102, regulations require monitoring of surface-mine sites during mining and, in some cases, after reclamation. The monitoring is normally focused on mine-drainage discharges to streams and springs. Regulations for mine drainage from surface mines in Pennsylvania are complex and exact limitations vary based on weather conditions ("dry" weather compared to 10-year, 24-hour precipitation events as listed in the Pennsylvania Code) and time period over which the limitations apply (30-day average, daily maximum, or instantaneous maximum) (Commonwealth of Pennsylvania, 2004). However, the most stringent limitation is that under "dry" weather conditions, the 30-day average effluent concentrations cannot exceed 3.0 mg/L for total iron, 2.0 mg/L for total manganese, and 35 mg/L for suspended solids, pH must be in the range of 6.0 to 9.0, and alkalinity must be greater than acidity.

In the Pennsylvania Code, Title 25, Subsection 87.116, regulations also state that ground-water quality, levels, infiltration rates, subsurface flows, and storage characteristics are to be monitored in a manner approved by the Pennsylvania Department of Environmental Protection (PaDEP) to determine effects of surface-mining activities on the quality and quantity of ground water, as well as on recharge capacity on reclaimed land, in the permitted mining area and adjacent areas (Commonwealth of Pennsylvania, 2004). The adjacent areas that are monitored usually are decided on a case-by-case basis. Several constituents are to be monitored for changes and must include specific conductance or total dissolved solids, pH, alkalinity, acidity, total iron, total manganese, sulfate, and water levels, but exact limitations for the constituents are not given in the regulations (Commonwealth of Pennsylvania, 2004). The limitations would have to be at least as stringent as the U.S. Environmental Protection Agency (USEPA) drinking-water standards (U.S. Environmental Protection Agency, 2004) if the affected aquifer or stream or spring were a source for a public water supply. However, in most cases, residents near surface mines use a well or spring for private domestic supply and these are not regulated.

Within the State of West Virginia, discharge of mine-related contaminants is governed by separate sets of regulations. For effluent discharged directly from pipes and other point sources, such as those commonly found draining tailings or treatment ponds, effluent concentrations are regulated by antidegradation standards established as part of recently enacted legislation (West Virginia Code 22-11-7b, Title 60 - Series 5, Antidegradation Implementation Procedures, established in 2001 [West Virginia Department of Environmental Protection, 2002]). Prior in-stream water-quality standards also apply and are designed to protect human health and animals, birds, fish, and other aquatic organisms and plant life. In-stream water-quality standards apply more to nonpoint sources of contamination and are governed by West Virginia Code 22-11-2, Title 46 - Series 1, Requirements Governing Water-Quality Standards (West Virginia Environmental Quality Board, 2005). The water-quality standards and the antidegradation standards vary depending on (1) drainage basin (stream) location and state-designated level of protection, and (2) conditions of weather, type of surface-water body, and time period over which standards apply. In general, the West Virginia in-stream standard for pH is similar to Pennsylvania (must be in 6.0 to 9.0 range), the iron maximum is 1.5 mg/L, and the manganese maximum is 1.0 mg/L. For ground-water quality in mining areas, a plan to monitor changes to ground-water baseline quality is required during mining operations and these regulations are described in the West Virginia Code 22-1-3, Title 38 - Series 3 (West Virginia Department of Environmental Protection, 2001).

Although the regulations state that contamination of ground-water quality is to be prevented, few studies (Rose and Cravotta, 1998; Williams and McElroy, 1995; McElroy, 1988; Friel and others, 1967; are examples) have examined whether ground water has been affected by surface coal mining near surface mining sites that have been reclaimed. Specifically, it has not been previously determined whether constituents that are indicative of mine drainage are present at concentrations greater than those for ground-water quality observed in unmined areas of the bituminous coal mining region of Pennsylvania and West

Virginia. Rose and Cravotta (1998) showed that in unmined areas, spring water and ground water typically were slightly acidic and contained much lower concentrations of dissolved solutes than water from mined areas. In mined areas, ground water ranged from alkaline to acidic and included elevated concentrations of sulfate, iron, manganese, and aluminum. In a study of water resources of Indiana County, Pa. (Williams and McElroy, 1995), ground water from the heavily mined geologic strata had higher median concentrations of iron, manganese, hardness, and total dissolved solids compared to that from the sparsely mined geologic strata. McElroy (1988), in a study of the ground-water resources of Fayette County, Pa., reported that wells near mining areas had statistically higher concentrations of sulfate, magnesium, and total dissolved solids than wells that did not penetrate a coal seam and that were more than 0.25 mi from mining sites. Friel and others (1967, p. 108) indicated that in the Monongahela Basin of West Virginia, median sulfate concentrations were greater in ground water from coal-bearing aquifers of Pennsylvanian age than from those aquifers that were in non-coal-bearing strata of Mississippian age. Ground water circulating through mined areas had increased acidity, total dissolved solids, and hardness, and often had dissolved metals carried in solution.

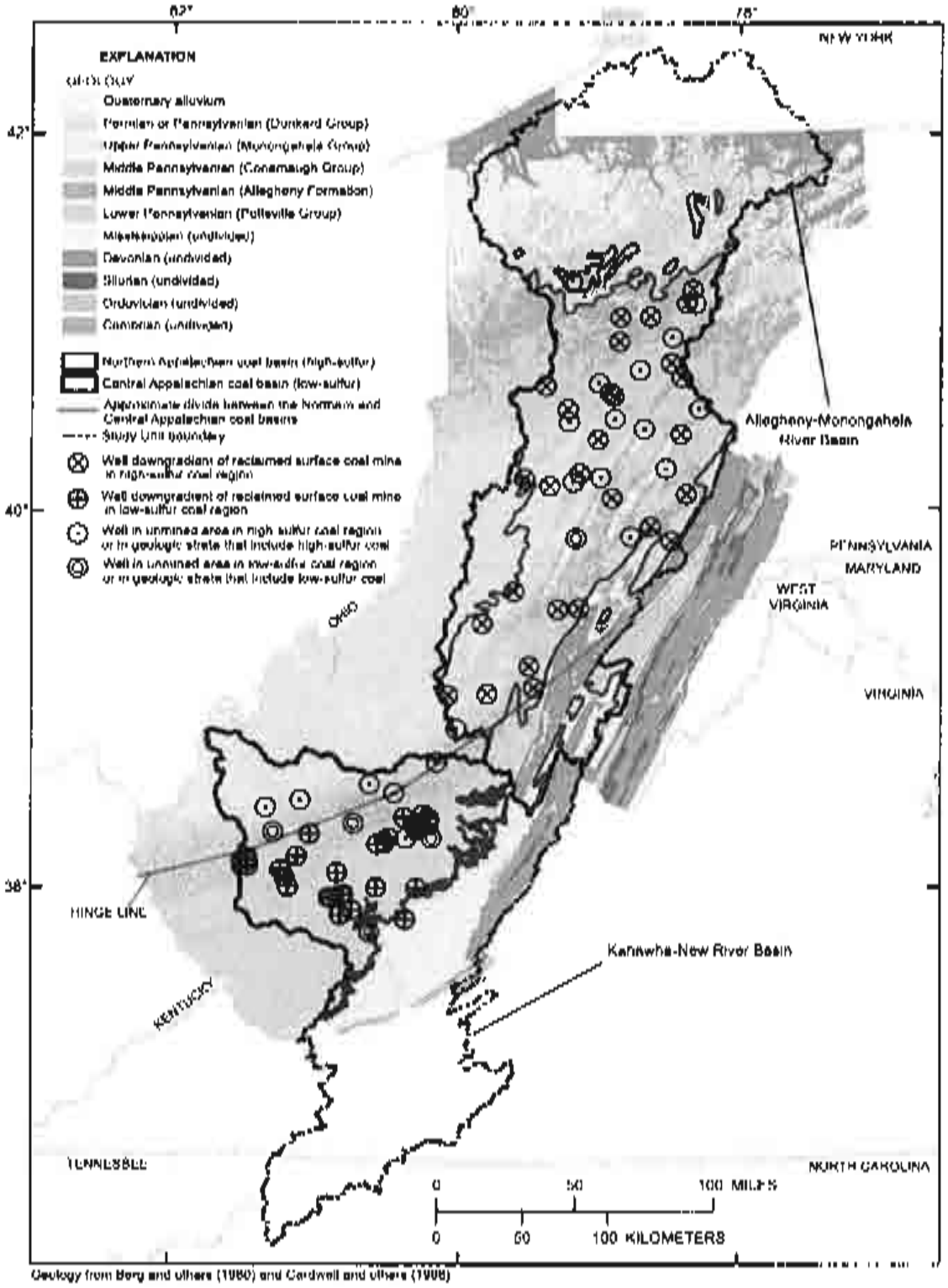
Purpose and Scope

This report summarizes major findings of a study of ground-water quality in surface coal-mining land-use areas. This study was conducted by the Kanawha-New River (KANA) NAWQA Program in the central Appalachian coal region of southern West Virginia (referred to as the low-sulfur coal region) and by the Allegheny-Monongahela River (ALMN) NAWQA Program in the northern Appalachian coal region in northern West Virginia, western Maryland, and southwestern Pennsylvania (referred to as the high-sulfur coal region) (fig. 1). The primary objective of the study was to compare and contrast ground-water quality from domestic-supply wells near reclaimed surface mines with water quality from equivalent wells in unmined areas. The report summarizes the (1) quality of ground water from unmined areas, (2) quality of water from aquifers immediately downgradient of reclaimed surface coal mines, (3) difference in quality of ground water from mined and unmined regions, (4) spatial extent of mine-drainage constituents in ground water from wells near reclaimed surface coal mines, (5) difference in quality of ground water between the high-sulfur and low-sulfur coal regions (in mined and unmined areas), and (6) difference in apparent age of ground water from mined and unmined areas and relation between apparent age and water quality in mined areas. The target population for sites to be sampled as part of this study was private wells (less than about 250 ft deep) downgradient (within about 2,000 ft) of surface mines that had been reclaimed (bond released) during 1985-1996. In 1997 to 1998, 58 such wells were sampled as part of this study, including 30 in the high-sulfur coal region and 28 in the low-sulfur coal region. In 1996 to 1998, 25 additional wells in unmined areas were sampled for comparison. All data used in this study were collected as part of the NAWQA Program. Data from previous studies were not used.

Description of the Study Area

This investigation considers ground-water resources in two adjoining drainage basins (fig. 1) of the Appalachian Plateaus Physiographic Province in New York, Pennsylvania, Maryland, West Virginia, Virginia, and North Carolina. The Appalachian Plateaus study area consists of rugged, deeply incised hilly terrain with uplifted plateaus capped by resistant layers of relatively flat-lying sandstone and shale (Fenneman, 1938; Fenneman and Johnson, 1946; U.S. Geological Survey, 1970). Geologic nomenclature used in this report is that of the Pennsylvania Topographic and Geologic Survey and the West Virginia Geological and Economic Survey. The study area is underlain by gently dipping, relatively flat lying, slightly folded sandstone, siltstone, shale, limestone, and coal of Devonian, Mississippian, Pennsylvanian, and Permian ages (fig. 1). Approximately 80 mineable coal seams are in the study area (Keystone Coal Industry Manual, 1994). The drainage basin in the northern part of the study area is the 8,778-mi^2 Allegheny-Monongahela River Basin in northern West Virginia, western Pennsylvania, and western Maryland. The basin is within the "northern Appalachian coal region" or the "high-sulfur coal region." Many researchers (Cecil and others, 1985; Donaldson and others, 1985; Maksimovic and Mowrey, 1993; Tully, 1996a, 1996b) have found that the bituminous coal, overburden rock, and underlying rock in much of the northern Appalachian coal region of western Pennsylvania and northern West Virginia generally has a high sulfur content. Cecil and others (1985) indicates that the high-sulfur coals are the group of coals with a median sulfur content greater than 1.0 percent in the northern Appalachian coal region. Tully (1996a) characterizes the sulfur content as greater than 1.5 percent. The drainage basin in the southern part of the study area is the 4,395-mi^2 Kanawha–New River Basin of southern West Virginia. Most of the southern basin is within the "central Appalachian coal region" or the "low-sulfur coal region." Cecil and others (1985) indicate that the low-sulfur coals are the group of coals with median sulfur content of less than 1.0 percent in the central Appalachian coal region. Tully (1996a) characterizes the sulfur content as less than 1.5 percent. Some of the northwestern Kanawha-New River drainage basin is within the high-sulfur coal region. In West Virginia, the high-sulfur and low-sulfur coal regions commonly are referred to as the northern and southern coal fields, respectively.

The differences in coal (and adjacent rock) sulfur content are due to depositional environments. The older rocks received more paleo-rainfall and paleo-recharge and thus the sulfur content was diluted compared to the younger rocks formed in a drier paleoclimate with higher ash, nutrient, and dissolved solids content that combine to result in higher sulfur content (Cecil and others, 1985). The younger rocks were deposited in marine



Geology from Berg and others (1980) and Cardwell and others (1968)

Figure 1. Study area, geology, and locations of wells sampled.
[The boundary between high-sulfur and low-sulfur coal regions coincides with that dividing the Northern and Central Appalachian Basins, respectively. The Pottsville Group in Pennsylvania (Berg and others, 1980) is recognized as the Kanawha, New River, and Pocahontas Formations in southern West Virginia (Cardwell and others, 1968).]

environments that had higher contents of sulfur and calcium carbonate. In contrast, the older rocks formed in areas that were marginally brackish, and so had lower content of sulfur and calcium carbonate. However, within most of the rocks of both ages, there are formations that represent marine, brackish, and marginally brackish paleoenvironments and each of these has high and low sulfur contents.

Cecil and others (1985) have indicated that the dividing line between the high-sulfur coal region and the low-sulfur coal region lies generally along the contact between the younger Pennsylvanian-aged rocks of the Allegheny, Conemaugh, and Monongahela Groups and the older Pennsylvanian-aged rocks of the Pottsville Group (Kanawha, New River, and Pocahontas Formations of West Virginia) in southern West Virginia (fig. 1). The dividing line between the high-sulfur coal region and the low-sulfur coal region commonly is referred to as the "hinge line." The "hinge line" is the same as the divide between the northern and central Appalachian coal basins shown in figure 1.

Average annual precipitation is about 42 in. for the central and northern Appalachian coal regions, although it is distributed unevenly with respect to topography and climatological regions (National Oceanic and Atmospheric Administration, 1997a, 1997b). Areas at higher elevations typically receive higher average amounts of precipitation than areas at lower elevations. Surface water is the principal source of water used in the region, accounting for 92 percent (2,340 Mgal/d) of all freshwater withdrawals (Pennsylvania and West Virginia Departments of Environmental Protection, written commun., 2000). Ground water accounts for only 8 percent (193 Mgal/d) of freshwater withdrawals. However, 12 percent of all water used for public water supply (51.9 Mgal/d) and 99.8 percent of self-supplied water used by rural homeowners (45.1 Mgal/d) is obtained from ground water. Land cover for the mining area is dominated by forested, agricultural, and rural residential uses (U.S. Geological Survey, 1992).

Geohydrologic Framework and Ground-Water Flow

The generalized hydrogeologic framework of the study area is based on regional topography, stratigraphy, and structure, with upland plateaus formed by resistant clastic rocks and valleys developed in zones of structural weakness. Structurally, the rocks in the Appalachian Plateaus study area are nearly flat to gently folded and commonly fractured. Pennsylvanian-age clastic rocks form the predominant outcropping units in the uplands and valley walls of southern and central West Virginia (Cardwell and others, 1968). Pennsylvanian and Permian stratigraphic units consist of sandstone, conglomerate, siltstone, shale, and coal, with local beds of limestone and dolomite. Where these bedrock units are not exposed at land surface, they commonly are overlain by a relatively thin layer of regolith or alluvium and, in northwestern Pennsylvania, by areally restricted, unconsolidated deposits of glacial origin (Berg and others, 1980).

Ground-water flow paths are relatively short and limited to two principal types of aquifer systems: (1) unconsolidated alluvial aquifers made up of sand, silt, clay, and gravel; and (2) fractured-bedrock aquifers made up of sedimentary and carbonate rocks (Puente, 1985). Because they tend to be shallow and thin, the alluvial aquifers present within the study area typically are not used as water supplies but can combine with soil and regolith to provide shallow storage for recent recharge. Fractured-bedrock aquifers are the primary aquifers in the study area. Locally, regolith, where alluvium is not present, is commonly thin with low permeability, providing little ground-water storage. Ground-water storage and flow in bedrock occurs through joints, fractures, and bedding-plane separations (Kozar and Mathes, 2001, p. 11). In the study area, secondary permeability due to jointing and stress-release fracturing accounts for most of the porosity and permeability in the bedrock, because original intergranular porosity commonly has been filled by calcium carbonate or silica cementation (Wyrick and Borchers, 1981).

Recharge to fractured-bedrock aquifers in the region occurs primarily as rainfall; snowmelt is only an important source of recharge in areas at elevations above 3,000 ft. Once precipitation falls on the surface, that part that does not run off to streams percolates into and through shallow soils and regolith and eventually recharges fractured-bedrock or alluvial aquifers.

A decrease in hydraulic conductivity with depth has been documented by several researchers. For a mine site in West Virginia, average hydraulic conductivity of an aquifer decreased from 10^{-5} m/sec at a depth of 150 ft to 10^{-9} m/sec at depths greater than 300 ft (Bruhn, 1985). For a site in Greene County, Pa., hydraulic conductivity decreased by an order of magnitude per 100 ft of depth to a depth of approximately 500 ft (Stoner, 1982). According to Callaghan and others (1998), the vast majority of ground-water circulation occurs at moderate depths of less than 300 ft.

In the Appalachian Plateaus, local fractured-bedrock aquifers generally of less than 5.0 mi^2 in area are defined by topographic valleys and boundary ridges. Each small valley may contain a locally distinct aquifer from which ground water discharges to a nearby stream or to deeper subregional or regional aquifers. The ridges surrounding the valley define the lateral boundaries of the local aquifer and its principal recharge area. Subregional aquifers (generally from 100 to 500 ft below land surface) occur at intermediate depths between the shallow local aquifer (less than 100 ft below land surface) and deeper regional aquifers. Subregional aquifers are larger than local aquifers and may include several smaller local aquifers. Discharge of ground water from subregional aquifers is primarily to tributary streams with drainage areas typically much larger than 5.0 mi^2. A small component of recharge to these aquifers is from deeper regional aquifers. These intermediate aquifers can extend beneath local topographic divides and discharge to regional aquifers. Depth to saline water (brines) has been used to infer the depth of regional aquifers (Callaghan and others, 1998). In the study area, the depth to brackish water ranges from a maximum in excess of 2,000 ft near the southern part of the

study area in West Virginia and in a minimum of less than 50 ft in the northwestern part of the study area in Pennsylvania (Fosse, 1986). Regional aquifers may contain saline or brackish water and typically discharge to large rivers.

Within the local aquifers, ground water typically flows from hilltops to valleys, perpendicular to local tributary streams, through an intricate network of strike-relief fractures and interconnected bedding-plane separations, commonly in a sawtooth pattern (fig. 2; Harlow and LeFan, 1990). Vertical hydraulic conductivity can be negligible, resulting in horizontal ground-water flow (especially within coal seams) that discharges as springs or seeps in valleys (Harlow and LeFan, 1990). A small proportion of the ground water flows deeper within the seams, out of the sawtooth or ridge, especially within coal seams and along bedding-plane separations, and likely reaches the valley to discharge locally to surface water or may recharge subregional and regional aquifers. Ground-water flow in valleys occurs primarily in bedding-plane separations beneath valley floors and in vertical and horizontal strike fractures along valley walls (Wyrick and Borchers, 1981). Enhanced permeability of bedrock in valleys may result in ground-water flow parallel to and beneath local tributary streams before ultimately discharging to surface water bodies.

The age of water in the subregional aquifer systems of the study area is not well documented. Based on ages of water in deeper wells sampled in West Virginia (Kozar, 1989), subregional ground water is at least 50 years old but may be much older, perhaps even centuries old. Deep aquifers in regional aquifers may be thousands of years old.

Surface coal mines can substantially change surface-water and ground-water flow patterns by altering recharge areas, recharge rates, and the porosity and permeability of aquifers.

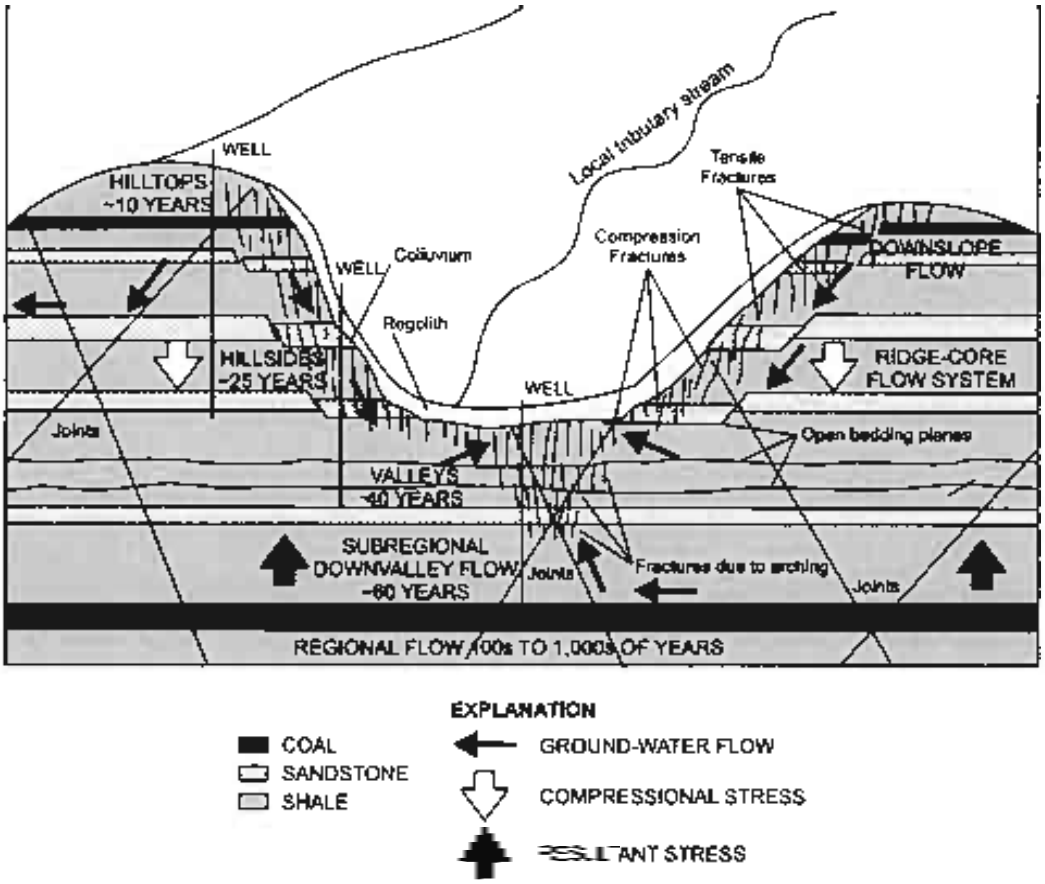


Figure 2. Conceptual model of ground-water flow in a fractured-bedrock aquifer in the Appalachian Plateau, including selected ground-water ages. Ground-water profile from Wyrick and Borchers, fig. 3.2-1, 1981 and Kozar, 1989. Ground-water ages presented in this figure are typical of ground water in the central Appalachian coal region and should not be considered characteristic of ground-water age in the northern Appalachian coal region.

Surface coal mines can be reclaimed to mimic the original contours of the land surface before mining (surface contour-surface mines). Ground-water flow in and through reclaimed contour-surface mines depends on many factors including the dip of bedrock at a mine, the composition and permeability of material within spoil, the age of spoil, and engineering controls such as ditches, terraces, sediment retention basins, ponds, and flumes or chutes. Each reclaimed surface mine has its own local ground- and surface-water flow system. There are, however, aspects common to most reclaimed contour-surface mines (Hawkins, 1998a, 1998b).

Typically, ground water will pool at the base of the surface-mine bench. Recharge to the ground water at a mine can originate as surface drainage through spoil and unmined areas upgradient of the backfilled contour bench (Hawkins, 1998a). If the coal seam dips away from the highwall, there may be little water stored within the spoil because ground water will freely flow down-dip through spoil. Where a coal seam dips towards a highwall, however, ground water commonly will accumulate and flow laterally along the base of the highwall until emerging to the surface as springs or seeps. Water emerging in seeps and springs may be conveyed away from a reclaimed contour-surface mine by ditches or drains to a nearby tributary stream and may be lost as a potential source of recharge. Mine spoil typically exhibits dual porosity, especially when more than one type of bedrock makes up the spoil (for example, sandstone, which usually consists of coarse-grained deposits such as sand, gravel, and boulders, and shale, which typically consists of fine-grained deposits within the spoil). Even where only one type of bedrock (for example, sandstone) is present within the spoil, differential settling of coarser-grained deposits (boulders and cobbles) is typical at the base of a cast spoil pile with finer-grained deposits (sand, dust, and gravel) near the surface during the backfilling process. Ground-water flow within the spoil is therefore highly variable because of interspersed fine- and coarse-grained deposits within the backfill.

Methods

Although a comparison of ground-water quality in mined areas or mined strata to sparsely mined areas or strata has been studied by other researchers as previously mentioned, this study collected data to examine ground-water quality in unmined areas and areas downgradient of only reclaimed surface coal mines. During June 1997-September 1998, ground water was sampled from 58 wells downgradient from reclaimed surface mines. In the low-sulfur coal region, 28 wells were sampled, and in the high-sulfur coal region, 30 wells were sampled (fig. 1). The wells sampled (Ward and others, 1998, 1999; Coll and Siwicki, 1997, 1999) are considered to be in the low-sulfur coal region if they are within the outcrop area of the Pocahontas, Kanawha, or New River Formation. All others are in the high-sulfur coal region. All wells were sampled using parts per billion, clean-hands protocols developed by the USGS NAWQA Program (Koterba and others, 1995). The downgradient wells were generally within about 2,000 ft from reclaimed surface mines, which were generally contour strip mines whose bonds had been released during 1985 to 1996. The 2- to 12-year bond-release criteria were selected because 2 years generally is considered to be the minimum time required for ground water to return to steady-state equilibrium after the reclamation process has been completed and 1985 was considered to be the year when provisions of SMCRA were fully implemented within the study area. Two of the downgradient wells were further than 2,000 ft from a mine. Shallow wells, generally less than 250 ft in depth, were also selected for sampling. In concept, the shallow, private, domestic supply wells will be more representative than the deeper wells in terms of intercepting recharge infiltrating from upgradient land surface, thus reflecting effects of the targeted land use (surface mining) on ground-water quality. A few wells were deeper than 250 ft. To identify position relative to mines, the principal direction of ground-water flow is assumed to coincide with the topographic gradient.

During July 1996 through September 1998, ground water also was sampled from 25 wells in areas that had not been mined, to serve as reference sites to the wells sampled near mines. Determination of a lack of coal mining for the wells in the unmined areas was made by examination of land-use geographic information system (GIS) coverages and by conducting well-owner interviews. In addition, a search was conducted for mining permit records near and upgradient of wells selected for sampling. The wells in unmined areas are in flow systems in fractured rock that are similar to the wells sampled near mines. Twenty of the wells sampled in unmined areas are in the high-sulfur coal region and 5 wells are in the low-sulfur coal region. Of the 25 reference sites sampled, 10 were within the Kanawha-New River Basin and 15 were in the Allegheny-Monongahela River Basin. Hereafter, the term "unmined" will be used instead of "reference."

The water samples from wells were analyzed for a broad range of constituents. Mine-drainage-related constituents included sulfate, pH, alkalinity, acidity, iron, manganese, aluminum, hardness, calcium, magnesium, specific conductance, total dissolved solids, and turbidity. Other constituents collected included physical field measurements (temperature and dissolved oxygen), other common ions (calcium, magnesium, sodium, and chloride), other trace metals, nutrients, dissolved organic carbon (DOC), dissolved gases, chlorofluorocarbons (CFCs), and radon. Calcium and magnesium are included with mine-drainage constituents in this report because the levels commonly are elevated because of alkaline treatment at mine sites. Bacteria samples were collected from the 28 wells sampled in the low-sulfur coal region but were not collected from the 30 wells sampled in the high-sulfur coal region. The wells sampled in the high-sulfur coal region were sampled for selected volatile organic compounds (VOCs) but wells sampled in the low-sulfur coal region were not because VOCs were not normally examined in mining-area studies in southern West Virginia. All samples, except CFCs and field measurements, were analyzed at the USGS National Water Quality Laboratory (NWQL). Results of all laboratory analyses are published in

Ward and others (1998, 1999) and Colland Siwicki (1997, 1999). References for methods for the analyses performed by the NWQL can be found in the aforementioned reports as well as in publications by researchers at the NWQL (U.S. Geological Survey, 2005). Methods for field measurements are described in the USGS National Field Manual for the Collection of Water Quality Data (U.S. Geological Survey, variously dated).

The acidity data in this report are calculated using the major components of acidity (pH and iron, manganese, and aluminum concentrations) in the method described in Rose and Cravotta (1998). Rose and Cravotta (1998) and Hedin and others (1994) have shown that calculated and measured acidities are comparable. Hedin and others (1994) found differences of less than 10 percent in measured and calculated acidities of mine-drainage samples where pH values were between 3.0 and 4.0. In alkaline samples, calculated acidity may exceed measured acidity because as the sample pH increases, iron and other metal ions form hydroxide complexes that have less capacity than the uncomplexed ions to consume bases (so there is less acidity contribution).

Ground-water age data may be used to (1) estimate average ground-water flow velocity, (2) determine relative ground-water flow directions, (3) better understand the ground-water flow processes within an aquifer, (4) assess when or if past and present land-use activities may affect an aquifer, and (5) develop strategies for protecting valuable ground-water reservoirs.

Estimates of ground-water age referred to within this report were determined by collecting ground-water samples and analyzing them for CFCs using analytical and interpretive methods of Busenburg and Plummer (1991, 1992) and Szabo and others (1996). CFCs are stable synthetic organic compounds that were first manufactured in the 1930s. CFCs, also known as Freons, are detectable at very low concentrations (one part in 10^{15} by weight or 1 pg/L in water) and are commonly used as ground-water tracers (Busenburg and Plummer, 1991). Concentrations of CFCs in ground water depend on their atmospheric partial pressures and the equilibrium temperature (recharge temperature) of the water at the base of the unsaturated zone during recharge (Busenburg and Plummer, 1992). Concentrations of CFC-11, CFC-12, and CFC-113 in the atmosphere have steadily increased since their development in the 1930s. Curves of atmospheric concentrations of CFC-11 and CFC-12 were reconstructed on the basis of production records (Chemical Manufacturer's Association, 1990) and rates of release to the atmosphere (McCarthy and others, 1977; Gamlen and others, 1986). Similar techniques were used to reconstruct atmospheric concentration curves for CFC-113 (Szabo and others, 1996). Assuming that CFC concentrations in recharge are proportional to atmospheric concentrations at a particular time and location, apparent ground-water age can be estimated to within 2 years from these curves by knowing the concentration of CFCs and the temperature of recharged water.

The estimation of apparent ground-water age with respect to mined and unmined areas and with respect to the high-sulfur northern and low-sulfur central coal regions is based on the assumption of piston flow. In a simple piston-flow system, water enters the aquifer in an upland area and flows downgradient to a discharge area with little or no mixing with water from areas other than the upland recharge area.

Quality-assurance (QA) samples were collected in addition to the environmental samples. Blanks were collected to assure that sampling procedures were not introducing bias into the data, and replicate and spiked samples were collected to verify that laboratory measurements were acceptable. Blank samples generally did not reveal problems with sampling procedures except for low levels of a few constituents (table 1). Analyses of replicates and field spike data did not indicate potential problems in analytical determinations at the laboratory.

Iron, aluminum, zinc, ammonia, and DOC were detected in several blank samples at concentrations greater than the minimum reporting limits of 3 µg/L, 1 µg/L, 1 µg/L, 0.015 mg/L, and 0.10 mg/L, respectively. Nevertheless, most detections of iron, zinc, and ammonia concentrations in the well samples were at concentrations greater than the level of contamination detected in the quality-assurance field blanks and thus are considered to represent valid data. However, concentrations of aluminum below 4.8 µg/L may have been affected by sampling or analytical bias. In addition, concentrations of DOC less than 16 mg/L that were reported for blanks may also have been affected by sampling or analytical bias. Because of the high concentration of DOC in blanks, DOC was included in tabular summaries of the data but not discussed further in the interpretation of water-quality data. Sampling and analytical bias were considered during statistical analyses of the available data. No VOCs analyzed in this study had field or trip blanks showing concentrations greater than an estimated 0.07 µg/L; therefore, censoring of VOC data was not warranted (table 1).

All instruments used for monitoring pH, specific conductance, water temperature, and dissolved oxygen were calibrated daily in the field. Review of log sheets for instrument calibration did not indicate any problems with field determinations of these constituents. Comparison of field and laboratory determinations of pH and specific conductance indicated close agreement for the wells sampled.

Concentrations of constituents sampled were compared to USEPA Drinking Water Standards, both primary and secondary. National Primary Drinking Water Regulations (primary standards) are legally enforceable standards that apply to public water supply systems (U.S. Environmental Protection Agency, 2004). Under the Federal Safe Drinking Water Act, a Maximum Contaminant Level (MCL)—the maximum allowable concentration—is set for inorganic and organic contaminants that can adversely affect public health. For comparison purposes, the water quality of private domestic wells near mined and unmined sites evaluated for this study was examined in relation to these standards. National Secondary Drinking Water Regulations (secondary standards) are nonenforceable guidelines established by the USEPA for contaminants such as sulfate, iron, and manganese that may cause water to have poor taste, odor, or color; may cause staining of plumbing fixtures; or may cause

Table 1. Summary of field and laboratory blank quality-assurance data with detections from groundwater samples collected as part of mining and non-use studies in the high-sulfur and low-sulfur coal regions.

[Concentrations or detections in micrograms per liter (µg/L), milligrams per liter (mg/L), nanograms per liter (ng/L), or nanograms per liter N (ng/L as N); NA, not applicable; <, less than; E, estimated]

Constituent name	Method detection limit	Unit of measure	Blank samples			Random samples from mined areas		
			Number of samples	Number of detections	Range of concentrations detected	Number of samples	Number of detections	Range of concentrations
Calcium, dissolved	0.02	mg/L	4	1	0.05	59	59	1.4–250
Magnesium, dissolved	0.1	mg/L	4	1	0.21	59	59	1.7–57.6
Silica, dissolved	0.1	mg/L	4	2	0.12–0.14	59	59	3.5–20.6
Ammonia, dissolved	1.0	µg/L	4	3	3.5–4.0	59	59	2.7–97.1
Iron, dissolved	3.0	µg/L	4	3	4.5–5.7	59	54	<3.0–51,000
Zinc, dissolved	1.0	µg/L	4	3	1.2–3.4	59	56	<1.5–47.2
Ammonia, dissolved as N	0.15	mg/L	3	3	0.015–0.051	59	53	<0.10–1.54
Dissolved organic carbon	0.5	mg/L	3	3	0.5–15.3	59	59	<0.5–11.2
Chloroform, total	0.02	µg/L	2	1	E0.01	59	4	0.006–0.118
Toluene, total	0.03	µg/L	2	1	E0.05	30	2	0.005–0.06
1,4-Dichlorobenzene, total	0.09	µg/L	2	1	E0.01	30	0	<0.09
Dichlorodifluoromethane, total	0.40	µg/L	2	1	E0.04	30	0	<0.40
1,2,4-Trimethylbenzene, total	0.06	µg/L	2	1	E0.07	30	13	0.006–0.142
m-p-Xylene, total	0.09	µg/L	2	1	E0.01	30	1	E0.008–<0.064

cosmetic effects such as tooth discoloration (U.S. Environmental Protection Agency, 2004).

A Mann-Whitney nonparametric rank-sum test was conducted to compare mine-drainage-related constituents and ground-water ages from samples downgradient of mined sites to those constituents from samples in unmined areas to determine if observed differences are statistically significant (Helsel and Hirsch, 1992, p. 118-124). Results of the tests are reported as significant at the 95-percent confidence level (p value <0.05). A nonparametric Kruskal-Wallis rank-sum test also was conducted to determine if mine-drainage constituents differ significantly based on orientation of sampled wells (in terms of dip and strike) relative to the mined area (Helsel and Hirsch, 1992, p. 159-165). A Spearman's rank correlation test was conducted to compare the age data to the distance and topographic data and to water-quality data, specifically the mine-drainage constituents listed previously (Helsel and Hirsch, 1992, p. 217-219). Multiple-step Kruskal-Wallis tests were conducted on ranks of the data to determine if median concentrations of mine-drainage constituents differ significantly in ground-water samples based on (1) location in coal-sulfur regions and (2) topographic location (Helsel and Hirsch, 1992, p. 159-163 and p. 206-207). For each of the two hypotheses tested, there were four groups including two for mining land use (mined, unmined) and two for either coal-sulfur region (high, low) or topographic location (hill, valley). The Kruskal-Wallis tests were conducted on the four sample groups, then on subsets of the four sample groups as needed to determine which sample group differed from the others. The significance level was alpha = 0.05 for comparison of four groups, and alpha = 0.025 for two groups. Nonparametric procedures were chosen because of unbalanced sample sizes and because of observations of skewness and of outliers in boxplots for many constituents. For purposes of ranking data with multiple minimum reporting levels for use in statistical tests, data were set equal to just less than the highest minimum reporting level. In boxplots, the highest minimum reporting level is shown as the censored level of the data.

Ground-Water Quality in Unmined Areas and Near Reclaimed Surface Coal Mines

Ground-water quality generally is influenced by several factors, such as climate, aquifer type and lithology, geologic structure, topographic setting, and land use. In this report, general comparisons of water quality including water types are presented. Background concentrations of constituents useful in characterizing mine drainage are assigned and discussed. Results of statistical testing to differentiate waters from wells in unmined areas and wells downgradient of areas with reclaimed surface coal mines (hereafter termed "mined areas") are examined for existence of contamination by mine drainage. The degree of the contamination compared to drinking-water-quality standards is discussed. The influence of hydrogeologic factors (spatial and temporal extents of changes downgradient of mined areas) as well as geological factors (changes in source geochemistry of aquifer rocks) also are examined.

Detailed data, except for CFC data, are not available in this report because they can be found in other publications (Coll and Siwicki, 1997, p. 188-201; 1999, p. 325-344; Ward and others, 1998, p. 342-351; 1999, p. 446-454). Data from USGS water-quality and ground-water databases also may be accessed at web site *http://waterdata.usgs.gov/*. Site identification numbers, local identification name, coal sulfur content region, land-use classification (unmined or mined), topographic location, distance downgradient from mine, and well depth are given in appendix 3.

Comparison of Water Quality Between Mined Areas and Unmined Areas

Surface mining of coal causes the disturbance of great volumes of coal, overburden rock, and pavement rock that underlies the coal. These disturbances may result in changes in hydrology of the area and expose previously buried rocks to oxygen and water that may result in changes in the quality of ground water beneath and downgradient from mined areas. Common effects include increased concentrations of dissolved sulfate, iron, aluminum, manganese, and other ions and metals that contribute to hardness and acidity and increased suspended solids and turbidity (Rose and Cravotta, 1998).

Data collected for this study are summarized in terms of the two most general data groups, ground water from wells in unmined and mined areas. However, to evaluate the data in a regional sense, the wells in unmined and mined groups were further subdivided into low-sulfur coal region and high-sulfur coal-region categories. This was done because characteristics of the coal-bearing rocks that form the aquifers, and thus affect ground-water quality, have been shown to have significantly different characteristics in previous research (Cecil and others, 1985; Donaldson and others, 1985; Maksimovic and Mowrey, 1993; Tully, 1996b). Wells in the northern Appalachian coal region of western Pennsylvania and northern West Virginia are assigned to the high-sulfur coal-region category. Wells in the central Appalachian coal region that includes southern West Virginia are assigned to the low-sulfur coal-region category. Cecil and others (1985) have indicated that the dividing line between the high-sulfur coal region and the low-sulfur coal region lies generally along the contact between the younger Pennsylvanian-aged rocks of the Allegheny, Conemaugh, and Monongahela Groups and the older Pennsylvanian-aged rocks of the Pottsville Group (Kanawha, New River, and Pocahontas Formations of West Virginia) in southern West Virginia (fig. 1).

Summary statistics describing ground-water chemistry for the unmined and the mined groups, including the minimum, maximum, median, and mean values for all constituents discussed in this report, are shown in table 2. The water-quality standards that will be compared to the concentrations in ground-water samples later in this section are listed in table 2.

Variations of Water Quality Related to Regional Differences in Coal Sulfur Content

The regional differences in major-ion composition of ground water in unmined and mined areas are illustrated using Piper diagrams (fig. 3). The principal composition of the ground water in unmined areas of the high-sulfur and low-sulfur coal regions is calcium-magnesium carbonate-bicarbonate type (69 percent of samples). The principal composition of ground water from mined areas in the high-sulfur and low-sulfur coal regions ranges from calcium-magnesium carbonate-bicarbonate type (52 percent of samples) to calcium-magnesium sulfate type (36 percent of samples).

The increased prevalence of sulfate-type waters in mined areas indicated that sulfate and other constituents would occur at different concentrations in mined areas compared to unmined areas. Box plots of several constituents (sulfate, iron, radon, pH, hardness, calcium, and specific conductance) are shown for high-sulfur versus low-sulfur coal regions in figure 4. Multiple stage Kruskal-Wallis tests (Helsel and Hirsch, 1992, p. 159-163, 200-207) on the four data subsets—mined high-sulfur, mined low-sulfur, unmined high-sulfur, and unmined low-sulfur—were conducted on the unbalanced ranks of the data, with results reported at a significance level of alpha = 0.025. Significantly different sample subset medians are denoted by different letters (A, B) on the box plots in figure 4. A subset median denoted AB in figure 4 is statistically the same as the median for code A and code B populations.

For unmined areas, there are no statistically significant differences between median concentrations of most mining-related constituents when high-sulfur and low-sulfur regions are compared. This generalization holds for sulfate, iron, pH, hardness, calcium, and specific conductance (fig. 4).

Medians of some constituents (specific conductance, calcium, radon, and pH) are greater in wells sampled in the high-sulfur mined areas compared to low-sulfur mined areas but not all are significantly greater. Iron and manganese (not shown in

Table 2. Summary statistics and related Federal drinking water standards for constituent concentrations in ground water from wells in mined (58 samples) and unmined areas (25 samples).

[mg/L, milligrams per liter; µg/L, micrograms per liter; pCi/L, picocuries per liter; µS/cm, microsiemens per centimeter; NTU, nephelometric turbidity units; <, less than; NA, not applicable; MCL, maximum contaminant level, enforceable standard; AMCL, alternate maximum contaminant level, enforceable standard; SDWR, secondary drinking water regulation—nonenforceable guideline for cosmetic or aesthetic effects; Advisory, nonenforceable advisory for taste or odor]

Constituent	Units	Samples from mined areas				Samples from unmined areas				U.S. Environmental Protection Agency drinking water standard[1]
		Minimum	Mean	Median	Maximum	Minimum	Mean	Median	Maximum	
Acidity	mg/L	0.03	6.22	1.46	45.7	0.037	10.9	1.74	86.3	NA
Alkalinity	mg/L	0	128	117	357	0	133	114	344	NA
Aluminum	µg/L	2.7	35.5	3.75	971	<1	8.04	3.1	79	50-200 (SDWR)
Ammonia	mg/L as N	<.015	.26	.141	1.54	<.015	.147	.078	.92	30 (Advisory)
Barium	µg/L	7.8	265	122	2,680	20	209	161	1,230	2,000 (MCL)
Beryllium	µg/L	<1.0	<1.0	<1.0	1.4	<1.0	<1.0	<1.0	<1.0	4 (MCL)
Calcium	mg/L	1.4	52.4	35.9	230	1.5	27.2	21	85	NA
Chloride	mg/L	.31	50.1	5.1	1,200	.9	21.6	7.3	240	250 (SDWR)
Chromium	µg/L	<1.0	2.23	1.9	8.3	<1.0	2.23	1.5	6.7	100 total (MCL)
Cobalt	µg/L	<1.0	6.55	<1.0	158	<1.0	1.73	<1.0	17	NA
Copper	µg/L	<1.0	6.25	<1.0	125	1.0	30.4	<1.0	409	1,000 (SDWR)
Dissolved organic carbon	mg/L	<.10	1.54	.8	10	.2	.76	6	2.5	NA
Dissolved oxygen[2]	mg/L	<.01	2.07	.4	11	<.01	1.98	2	7.3	NA
Fluoride	mg/L	1	.21	.14	1.6	.1	.272	.2	1.1	4 (MCL)
Hardness	mg/L	13	188	135	820	3	95.8	70	290	NA
Iron	µg/L	<3.0	4,440	840	31,000	<3.0	733	80	5,170	300 (SDWR)
Lead	µg/L	<1.0	<1.0	<1.0	19	<1.0	<1.0	<1.0	29	15 (USEPA action level)
Magnesium	mg/L	1.7	13.9	9.9	57	1.1	6.78	4.4	19	NA
Manganese	µg/L	<1.0	527	235	2,780	<1.0	249	94	1,530	50 (SDWR)
Net alkalinity	mg/L	-86.3	116	111	355	-9.3	130.6	113	344	NA
Nickel	µg/L	<1.0	8.7	1.3	166	<1.0	2.1	<1.0	11	NA
Nitrate plus nitrite	mg/L as N	<.050	.23	.05	2.9	<.050	1.02	.07	14	10 (MCL)
Nitrite	mg/L as N	<.010	.011	.01	.021	<.010	.054	.01	1.1	1 (MCL)
pH	standard units	3	6.51	6.7	8.8	4.2	6.85	6.9	9.2	6.5-8.5 (SDWR)
Potassium	mg/L	.52	2.17	1.75	7.5	.12	1.62	1.4	4.5	NA
Radon	pCi/L	26	306	181	4,132	43	812	380	4,688	300 (MCL)/4,000 (AMCL)
Silica	mg/L	3.5	11.1	11	20	5.9	9.37	8.8	15	NA
Sodium	mg/L	.52	43.9	9.35	806	.5	35.8	18	270	30-60 (Advisory)
Specific conductance	µS/cm at 25°C	42	559	358	4,210	78	338	285	1,350	NA
Sulfate	mg/L	<.10	97.1	32.5	700	.45	14.75	14	53	250 (SDWR)
Total dissolved solids	mg/L	32	362	218	2,300	34	207	180	773	500 SDWR)
Turbidity	NTU	.25	57.7	6	1,000	<.01	24.9	2	500	NA
Zinc	µg/L	<1.0	33.9	8.1	472	<1.0	12.9	3.4	85	5 (SDWR)

[1] These standards only regulate public supplies and do not apply to private domestic wells.

[2] Only 56 and 24 samples in mined and unmined areas, respectively.

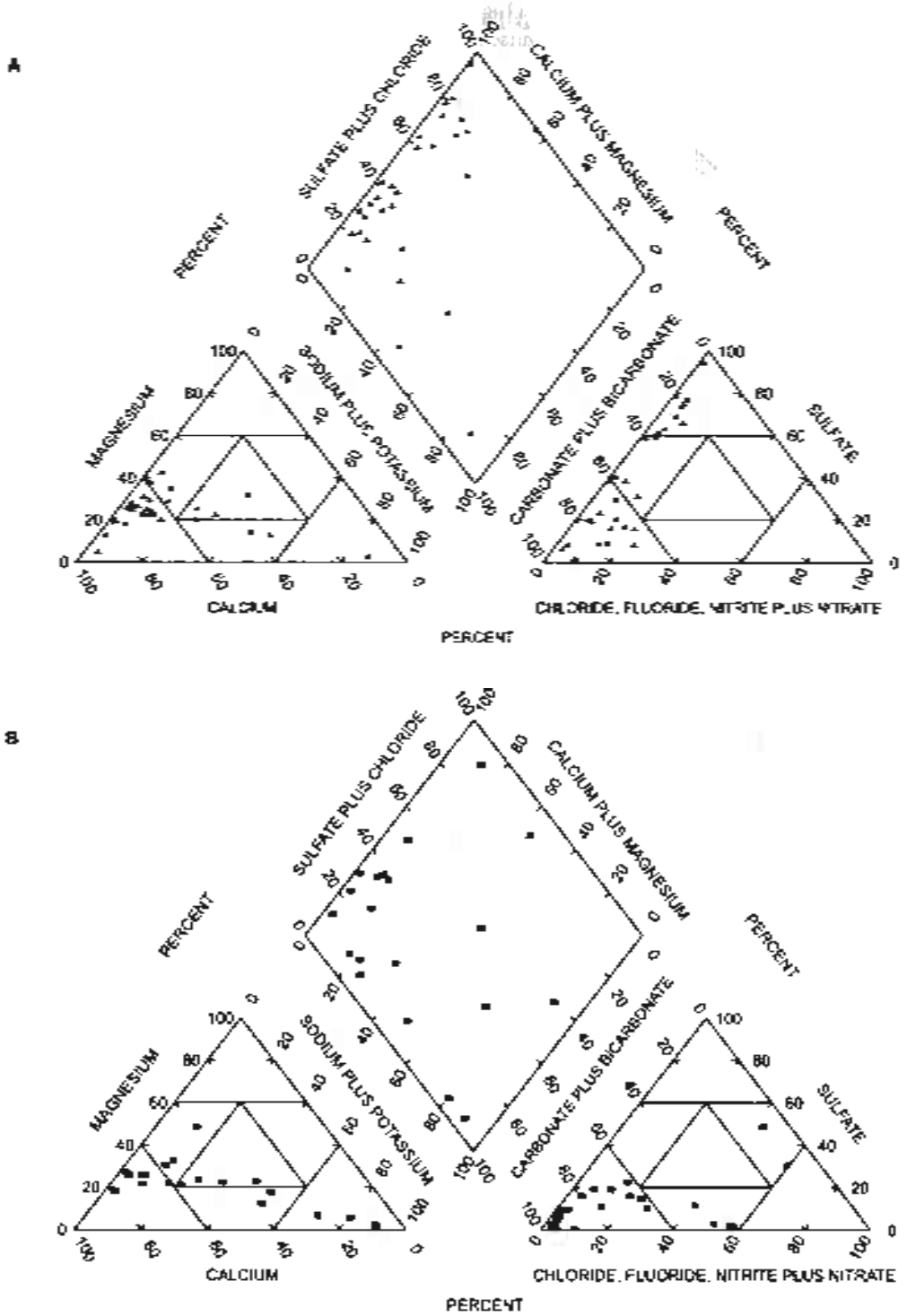


Figure 3. Major-ion composition of ground water (A) downgradient of reclaimed surface coal mines in the high-sulfur coal region, (B) in unmined areas in the high-sulfur coal region, (C) downgradient of reclaimed surface coal mines in the low-sulfur coal region, and (D) in unmined areas in the low-sulfur coal region.

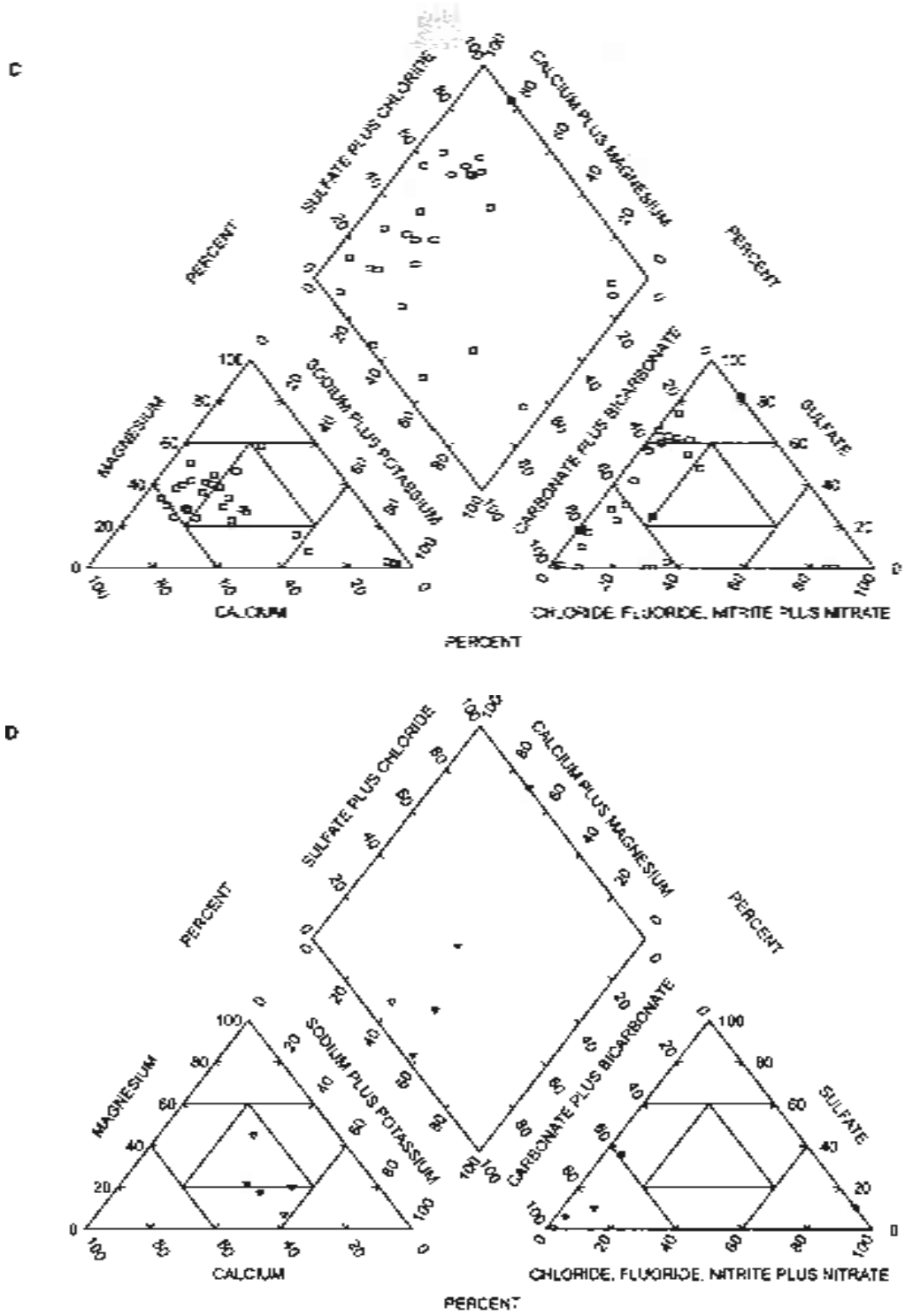


Figure 3. Major-ion composition of ground water (A) downgradient of reclaimed surface coal mines in the high-sulfur coal region, (B) in unmined areas in the high-sulfur coal region, (C) downgradient of reclaimed surface coal mines in the low-sulfur coal region, and (D) in unmined areas in the low-sulfur coal region.—Continued

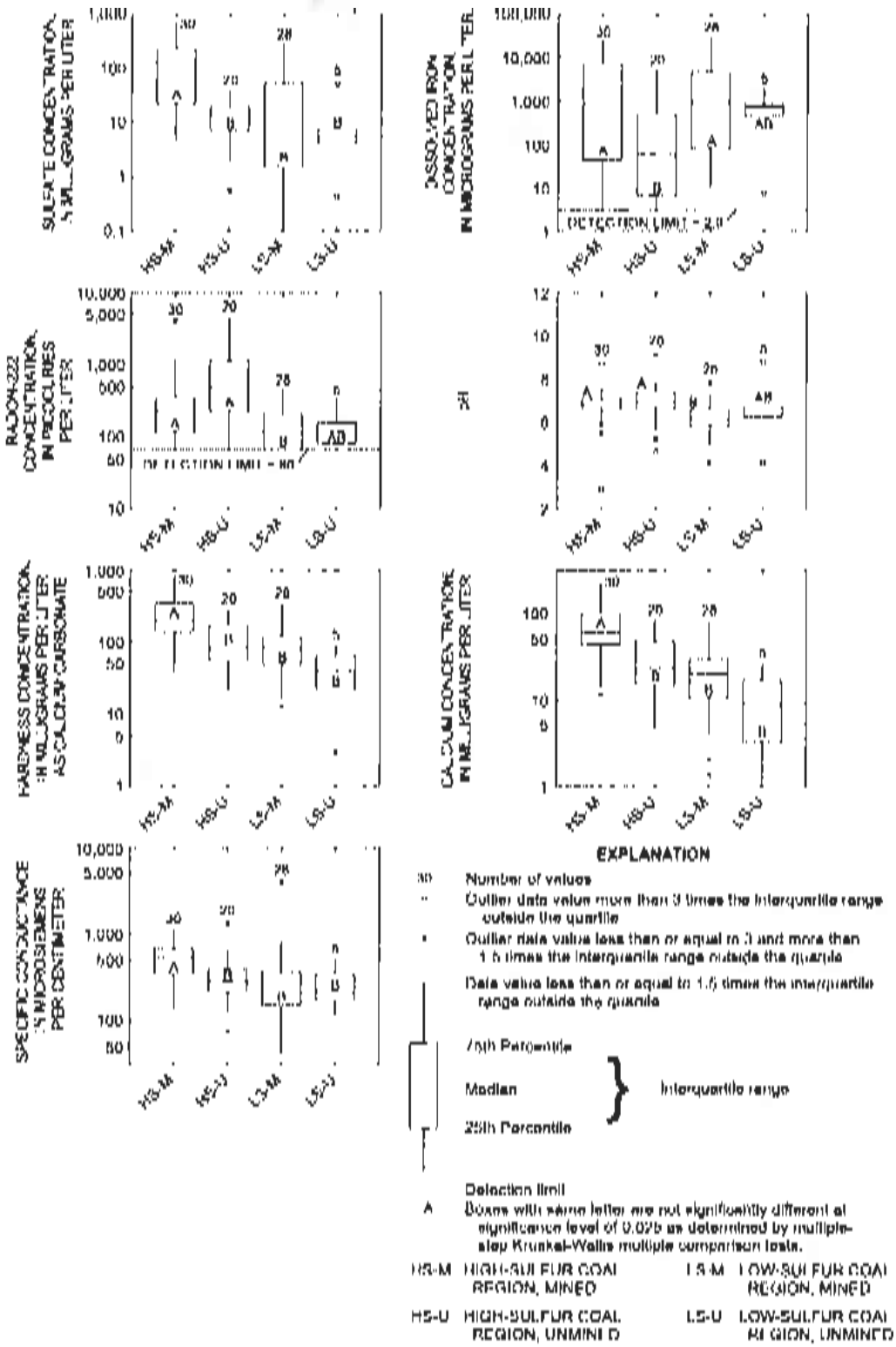


Figure 4. Selected mine-drainage constituents in ground water downgradient of reclaimed surface coal mines and in unmined areas of high-sulfur and low-sulfur coal regions.

figure 4) concentrations are not significantly different between high-sulfur and low-sulfur mined areas. Median concentrations of sulfate, hardness, specific conductance, and calcium are significantly greater for mined areas in the high-sulfur coal region compared to mined areas in the low-sulfur coal region, and to any unmined location. Radon and pH are significantly greater in the high-sulfur coal region for mined areas compared to mined areas in the low-sulfur coal region. The radon and pH in any unmined area was not different from mined areas in the high-sulfur coal region. Iron and other mine-drainage constituents showed no significant difference related to coal-sulfur region.

Curiously, acidity is not significantly greater in any one region or mining condition. This may suggest that reclamation practices are effectively reducing acid concentration. Calcareous materials used as amendments to mine spoil and high-wall fill are likely helping to reduce acid through neutralization reactions that increase the calcium and magnesium levels. Many more mining operations also now employ special handling methods for highly pyritic materials, thus further reducing release of acid into the ground water. Calcareous minerals are also more abundant in general in rocks of the high-sulfur area.

To summarize, there are some significant differences in mine-drainage contaminant levels between high-sulfur and low-sulfur regions but the greatest differences are between mined high-sulfur region sites and all other sites because the combination of source geochemical factors and mining disturbance are maximized. Iron levels are also increased in mined areas regardless of coal-sulfur content. Because of the subtle differences in ground-water quality when evaluated by coal-sulfur region, hereafter, the focus will be on comparing the results from unmined areas with results from mined areas regardless of coal-sulfur content.

Background Water Quality in Unmined Areas

To identify water affected by mine drainage in general and to later estimate the extent of lateral and vertical migration of waters originating from reclaimed surface mines, a background concentration representing unaffected waters was determined for selected mine-drainage constituents — sulfate, iron, manganese, aluminum, zinc, hardness, and total dissolved solids that generally are expected to be increased in mined areas (Rose and Cravotta, 1998). Alkalinity, acidity and pH background concentrations were not calculated because these constituents could be either higher or lower in mined areas compared to unmined areas depending on whether acidic mine drainage or neutral or alkaline mine drainage occurs (Rose and Cravotta, 1998). Data from the wells representing aquifers in unmined areas were used. Concentrations were ranked, and the concentration that was exceeded in 10 percent of the samples (the 90th percentile value) for each constituent was used to represent the background concentration.

Using sulfate as an example, only one sample from an unmined area had a sulfate concentration greater than 50 mg/L. Both the mean and median sulfate concentrations in ground water from wells representing unmined areas are about 14 mg/L. Using the aforementioned procedure, the background concentration (90th percentile value) was determined to be 26 mg/L. This value is low compared to those determined by other researchers. McCurry and Rauch (1986) working in northern West Virginia, Brady (1998) in Pennsylvania, and river assessments across Western Pennsylvania by Sams and Beer (2000) have yielded higher background concentrations for sulfate of 50, 40, and 75 mg/L, respectively. Background concentrations for selected constituents are plotted in subsequent illustrations to facilitate identification of concentrations elevated above background levels.

Major Water Characteristics and Mine-Drainage Constituents

Ground water in mined areas appears to have water quality distinct from unmined areas. Ground-water chemistry in mined areas is characterized by elevated specific conductance and total dissolved solids associated with mine-drainage constituents such as sulfate, dissolved iron, dissolved manganese, dissolved aluminum, and hardness (table 2). Median pH values of 6.7 and 6.9 characterize the mined and unmined areas, respectively. Alkalinities were about the same in mined and unmined areas. Box plots of 10 selected constituents illustrate the frequency distributions of data for mined and unmined areas (fig. 5). Background concentrations also are shown in figure 5. To determine if medians of the two sample distributions plotted in figure 5 were significantly different, Mann-Whitney rank-sum tests were conducted. The results of the tests are summarized for 30 constituents in table 3.

Median concentrations of sulfate, iron, manganese, aluminum, zinc, calcium, magnesium, hardness, specific conductance, turbidity, potassium, silica, and ammonia were significantly higher in mined areas compared to unmined areas, whereas radon and fluoride were significantly lower in mined areas compared to unmined areas. The pH is significantly lower in mined areas compared to unmined areas (fig. 5).

Trace Elements

Sixteen trace elements including aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, lead, molybdenum, nickel, selenium, silver, uranium, and zinc were determined by analysis of ground-water samples. Results were highly censored, with many concentrations reported to be less than reporting limits of the analytical methods. Data for all 16 elements are published elsewhere (Coll and Siwicki, 1997, p. 197-198; 1999, p. 337-340; Ward and others, 1998, 1999) and summary statistics for concentrations of only 9 elements are presented in table 2. Elements related to coal mining, such as selenium, that had no concentrations at or above the method reporting limit of 1 µg/L, are not discussed hereafter.

Table 3. Mann-Whitney rank-sum test results comparing median concentrations of constituents in ground water from wells in mined and unmined areas.

[Shaded cells are significantly different with a p value of < 0.05; mg/L, milligrams per liter; µg/L, micrograms per liter; NTU, nephelometric turbidity units; pCi/L, picocuries per liter; µS/cm, microsiemens per centimeter]

Chemical constituent (units)	Median		p value
	Mined (total = 58)	Unmined (total = 25)	
Acidity (mg/L)	1.46	1.74	0.709
Alkalinity (mg/L)	117	114	.551
Aluminum (µg/L)	3.75	3.1	.0236
Ammonia (mg/L)	.141	.078	.043
Barium (µg/L)	122	161	.309
Calcium (mg/L)	35.5	21	.0207
Chloride (mg/L)	5.1	7.3	.321
Chromium (µg/L)	1.9	1.5	.478
Cobalt (µg/L)	<1	<1	.0964
Copper (µg/L)	<1	<1	.961
Dissolved organic carbon (mg/L)	.8	.6	.03
Dissolved oxygen	.4	.2	.56
Fluoride (mg/L)	.14	.2	.0448
Hardness (mg/L)	135	70	.0082
Iron (µg/L)	840	80	.0031
Magnesium (mg/L)	9.9	4.4	.0026
Manganese (µg/L)	235	94	.0221
Net alkalinity (mg/L)	111	113	.785
Nickel (µg/L)	1.3	<1	.146
Nitrate plus nitrite (mg/L as N)	.05	.07	.249
pH (standard units)	6.7	6.9	.0453
Potassium (mg/L)	1.75	1.4	.0419
Radon (pCi/L)	181	380	.0092
Silica (mg/L)	11	8.8	.0494
Sodium (mg/L)	9.35	18	.297
Specific conductance (µS/cm at 25°C)	358	285	.028
Sulfate (mg/L)	32.5	14	.0024
Total dissolved solids (mg/L)	218	180	.115
Turbidity (NTU)	6	2	.0047
Zinc (µg/L)	8.1	3.4	.0393

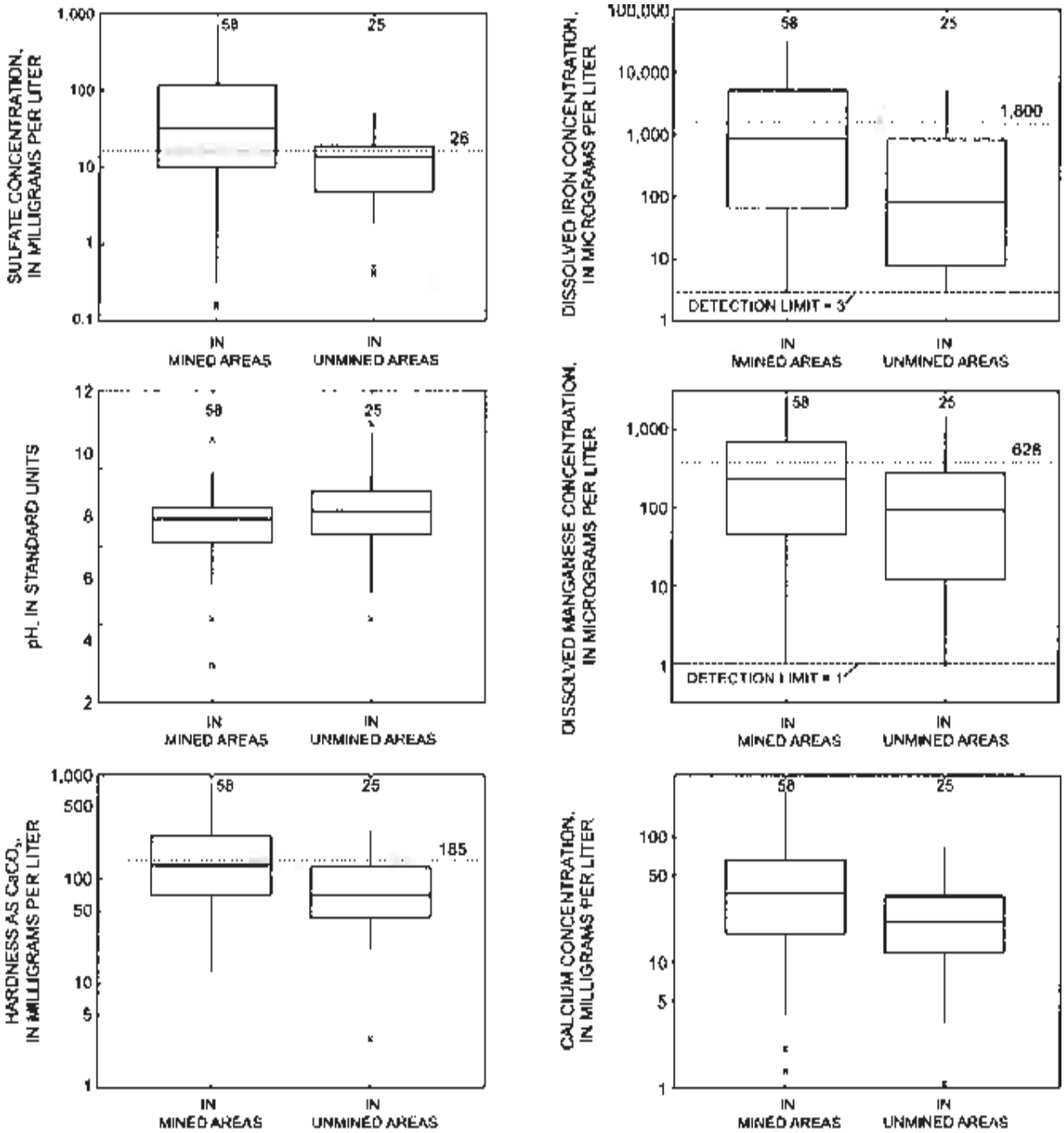


Figure 5. Concentration distribution of mine-drainage constituents downgradient of reclaimed surface coal mines and in unmined areas.

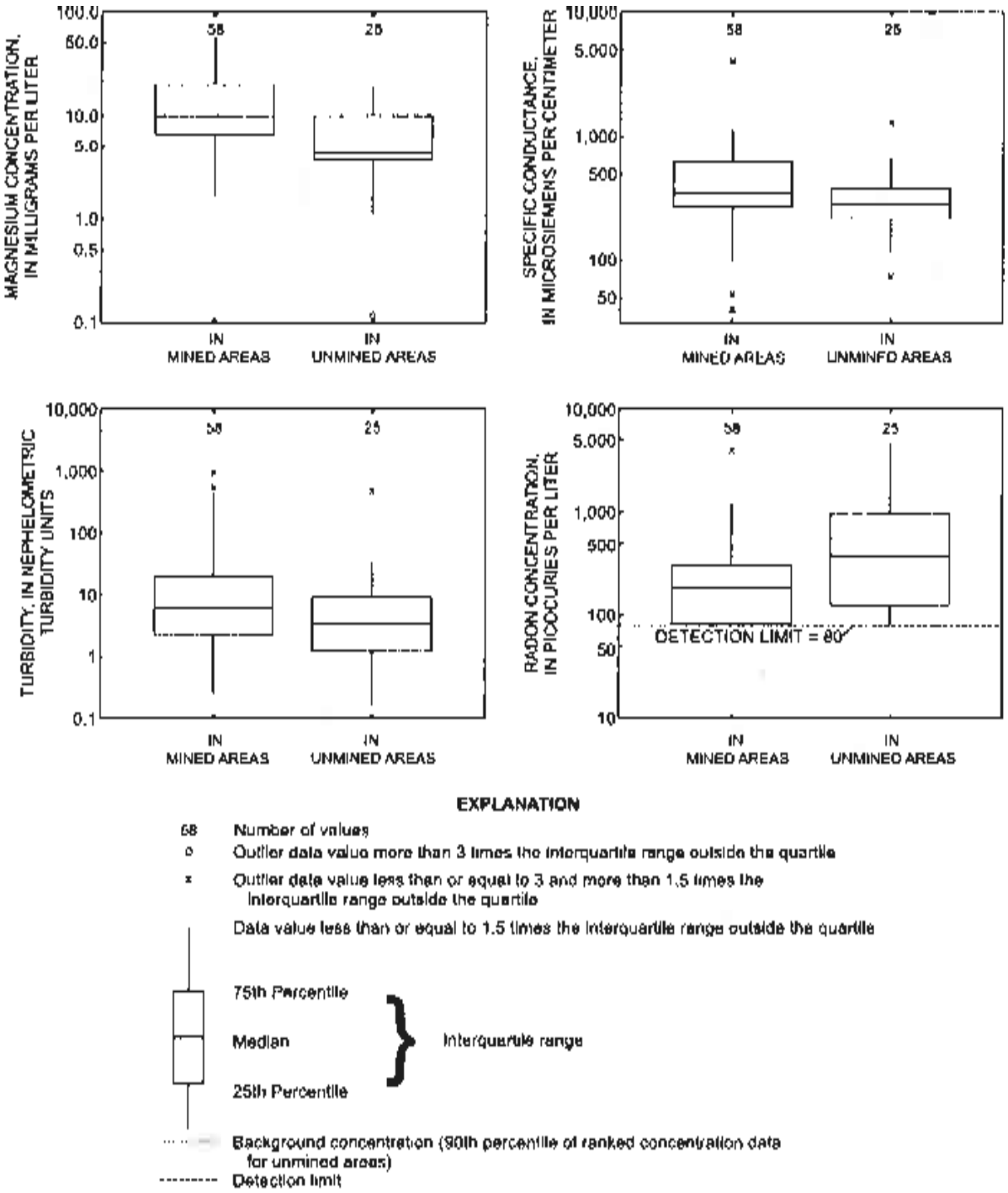


Figure 5. Concentration distribution of mine-drainage constituents downgradient of reclaimed surface coal mines and in unmined areas.—Continued

Six elements for which there was at least one concentration above the minimum reporting limit were analyzed for significant differences between mined and unmined areas. Medians of aluminum, chromium, and zinc are higher for mined areas than in unmined areas (table 2). Also, greater concentrations for outlier values and greater numbers of outlier points for aluminum and zinc were measured in mined areas compared to unmined areas (fig. 6). Results of Mann-Whitney rank-sum tests indicate that only zinc is significantly greater in mined areas (median of 8.1 µg/L) than in unmined areas (median of 3.4 µg/L) (fig. 6 and table 3). The results for aluminum indicate a significantly greater median of 3.75 µg/L in mined areas compared to a median of 3.1 µg/L in unmined areas; however, these results (table 3 and fig. 6) are inconclusive due to the bias resulting from low-concentration detections of aluminum in analyses of three of four blank samples. Medians for other trace elements were not significantly different between mined and unmined areas.

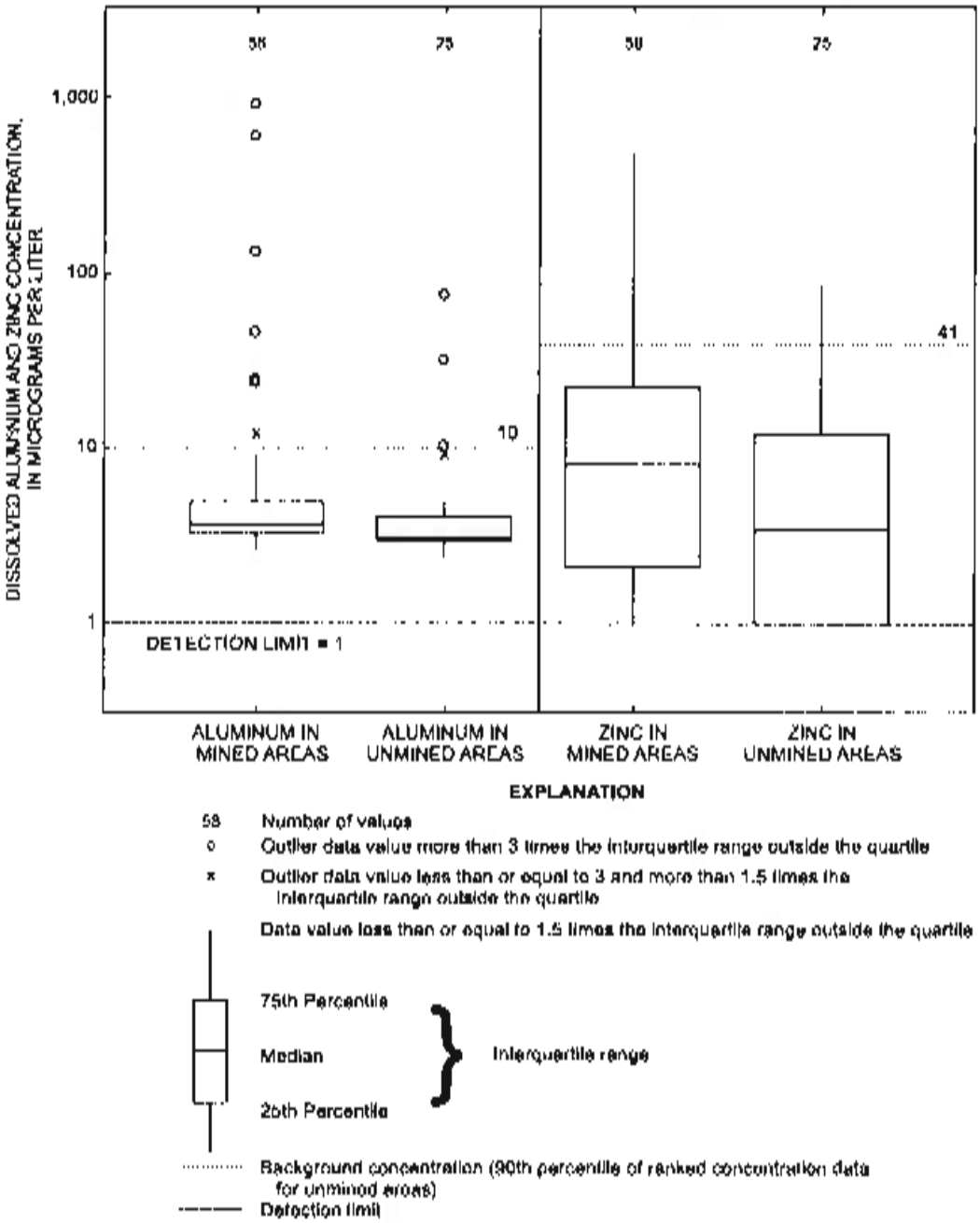


Figure 6. Concentration distribution of trace elements that are significantly different between mined and unmined areas.

Volatile Organic Compounds

Samples for analysis of volatile organic compounds (VOCs) were collected from 30 wells in mined areas and from 15 wells in unmined areas in the northern Appalachian coal region. Analytical reporting limits (detection limits) varied widely among the compounds. Twenty-seven of the 30 samples from mined areas and each of the 15 samples from unmined areas had at least one VOC concentration above the minimum reporting limits. Of the 86 compounds analyzed, at least 1 detection was reported for 23 compounds. Seven compounds were detected in samples collected in mined areas only, and eight compounds were detected in samples collected from unmined areas. The compounds detected, the maximum concentrations reported, and the MCLs are listed in table 4.

Of the 23 detected compounds, 7 are compounds that include benzene, toluene, and xylene (BTEX compounds), 11 are halogenated straight chain compounds (alkanes and alkenes), 2 are ketones, 1 is an ether, and the remaining 2 compounds (tetrahydrofuran and carbon disulfide) are unrelated by functional group. In samples collected from mined areas, there were 33 detects for BTEX compounds (1.10 detects per sample) and 17 detects for halogenated straight chain compounds (0.57 detects per sample). In samples collected from unmined areas,

Table 4. Maximum concentration detected and number of samples with detections of volatile organic compounds (VOCs) for mined and unmined areas in the northern Appalachian coal region.

[E, estimated; <, less than; --, no MCL has been set; *, Total for trihalomethanes is 0.08 mg/L; P, Regulation is pending]

Compound name	Minimum reporting limit (micrograms per liter)	U.S. Environmental Protection Agency maximum contaminant level[1] (micrograms per liter)	Samples from wells in mined areas		Samples from wells in unmined areas	
			Maximum concentration (micrograms per liter)	Number of detections (total = 30)	Maximum concentration (micrograms per liter)	Number of detections (total = 15)
Bromodichloromethane	0.048	80P*	< 0.048	0	0.05	1
Tetrachloromethane	.088		E .01	1	< .088	0
Dibromochloromethane	.18	80P*	< .18	0	E .03	1
Chloroform	.052	80P*	.12	8	.20	6
Toluene	.05	1,000	E .04	3	.067	1
Benzene	.10	5	.598	6	E .02	1
Ethylbenzene	.030	700	E .01	1	< .030	0
Chloromethane	.25	--	E .10	3	E .029	2
Dichloromethane	.38	5	< .38	0	.382	7
Tetrachloroethylene	.10	5	< .10	0	E .04	1
1,1,1-Trichloroethane	.032	200	E .02	1	.06	1
Trichloroethylene	.038	5	< .038	0	.17	1
Carbon disulfide	.37	--	E .318	7	E .01	2
cis-1,2 Dichloroethylene	.038	70	< .038	0	.10	1
Styrene	.042	100	E .040	1	E .005	1
1,2,3-Trimethylbenzene	.12		< .12	0	E .002	1
1,2,4-Trimethylbenzene	.056	-	.140	18	E .041	10
Methyl iodide	.021		.080	3	< .021	0
Methyl *tert*-butyl ether	.17	--	E .030	1	< .17	0
4-Methyl-2-pentanone	.37	--	E .10	1	< .37	0
2-Butanone	1.6	--	< 1.6	0	E 1.1	1
Tetrahydrofuran	9.0	--	E 3.0	7	< 9.0	0
m- and p-Xylene	.06	10,000	E .008	1	< .06	0

[1]These maximum contaminant levels only regulate public supplies and do not apply to private domestic wells.

there were 14 detects of BTEX compounds (0.93 detects per sample) and 21 detects of the straight chain compounds (1.4 detects per sample). Detection frequencies of benzene and toluene are greater in mined areas (20 and 10 percent) than in unmined areas (7 and 7 percent) (fig. 7).

The greater frequency of detection of VOCs in mined areas could be related to accidental releases of petroleum products from machinery used during mining; however, it also is possible that atmospheric deposition is a pathway by which VOCs reach ground water at the low concentration levels measured in this study (U.S. Environmental Protection Agency, 1999). The occurrence of VOCs at low levels in the atmosphere can be widespread because of emissions from automobiles, industries, heating, power generating plants, and other sources (National Science and Technology Council, 1997).

Relation to Drinking Water Standards

USEPA primary standards for barium and lead were exceeded at one site each in mined areas. USEPA primary standards for nitrate, nitrite, and lead were exceeded at one site each in unmined areas. However, because none of these constituents are commonly related to coal mining, they are not further discussed.

USEPA Secondary Maximum Contaminant Levels (SMCL) for aluminum, chloride, iron, manganese, pH, sulfate, and total dissolved solids were exceeded at two or more sites in mined areas (table 5). None of the samples from unmined areas exceeded the aluminum, chloride, or sulfate SMCLs. Generally, the constituents related to mining (sulfate, aluminum, iron, manganese, pH, and total dissolved solids) exceeded SMCLs more frequently in mined areas compared to unmined areas. The concentration of sulfate exceeded the SMCL of 250 mg/L in six samples from mined areas. Concentrations of chloride exceeded the SMCL of 250 mg/L in only two samples from mined areas. The SMCL of 300 µg/L for iron was exceeded in samples from 64 percent (37 of 58) of sites in mined areas and 36 percent (9 of 25) of sites in unmined areas. The SMCL of 50 µg/L for manganese was exceeded in samples from 71 percent (41 of 58) of sites in mined areas and 56 percent (14 of 25) of sites in unmined areas. The pH of samples from 38 percent (22 of 58) of sites in mined areas and 36 percent (9 of 25) of sites in unmined areas was outside of the Secondary Drinking Water Standard range of 6.5 to 8.5. The concentration of aluminum exceeded the SMCL range maximum value of 200 µg/L in only three samples from mined areas. The concentration of total dissolved solids exceeded the SMCL of 500 mg/L in samples from 16 percent (9 of 58) of sites in mined areas and 4 percent (1 of 25) of sites in unmined areas. All VOC concentrations reported were below the USEPA MCL values for drinking-water supplies.

The ground water in mined areas has concentrations of constituents indicative of pyrite oxidation and treatment of mine drainage with calcareous amendments. Higher concentrations of calcium, magnesium, and hardness can result from leaching of minerals from rocks due to reactions with sulfuric acid, from excavation of calcareous overburden to obtain more calcareous materials to use in filling high wall areas to neutralize AMD (Hawkins, 1998b, p. 17-1–17-6, Brady, 1998, p. 9-7–9-8), or from calcareous amendments introduced at the site to neutralize AMD in the mine spoil and at locations below seeps (Brady, 1998, p. 10-6–10-8). The calcareous materials used at surface-mine areas would also increase pH and decrease acidity and may explain why acidity was not significantly different and why the pH difference, though statistically significant, was small between ground water in mined and unmined areas. Total dissolved solids, specific conductance, and turbidity usually will increase as rocks are disturbed and broken and more minerals dissolved through exposure and weathering. Radon usually will decrease (resulting in less radon available for dissolution into the ground water) because it is released to the atmosphere by increased fracturing of rocks above and below the surface.

Table 5. Maximum values reported and number of samples exceeding the U.S. Environmental Protection Agency Secondary Maximum Contaminant Level for aluminum, chloride, iron, manganese, pH, sulfate, and total dissolved solids in ground water downgradient of reclaimed surface coal mines and wells in unmined areas.

[SMCL, Secondary Maximum Contaminant Level, µg/L, micrograms per liter, mg/L, milligrams per liter]

Contaminant (units)	SMCL	Samples from wells in mined areas		Samples from wells in unmined areas	
		Maximum concentration	Number of samples exceeding SMCL (total = 58)	Maximum concentration	Number of samples exceeding SMCL (total = 25)
Aluminum (µg/L)	200	971	3	79	0
Chloride (mg/L)	250	1,200	2	240	0
Iron (µg/L)	300	31,000	37	5,170	9
Manganese (µg/L)	50	2,780	41	1,530	14
pH (standard units)	6.5-8.5	3.0-8.8	22	4.2-9.2	9
Sulfate (mg/L)	250	700	6	53	0
Total dissolved solids (mg/L)	500	2,300	9	773	1

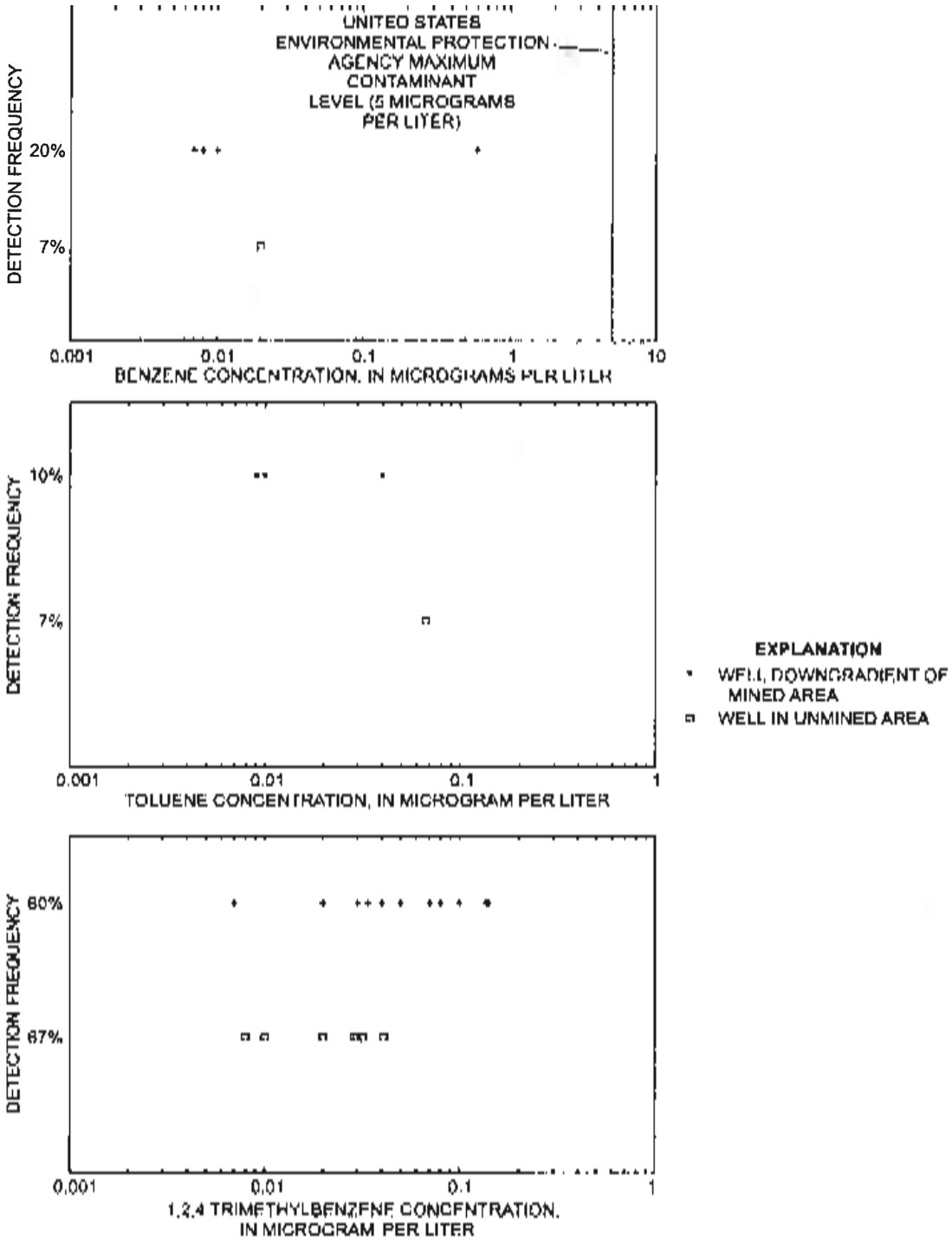


Figure 7. Concentration distribution of selected volatile organic compounds related to petroleum compounds in ground water downgradient of reclaimed surface coal mines and in unmined areas.

Spatial and Temporal Relations and Local Extent of Mine-Drainage Constituents in Mined Areas

Ground-water quality is affected by mine drainage to various degrees at various distances from reclaimed surface mines. There also are changes in the quality of ground water as a function of depth, topography, coal-sulfur content, and time, assuming that 1) movement of ground water is by piston flow, 2) flow paths coincide with topographic downgradient directions, 3) the primary discharge areas are surface-water bodies, and 4) springs do not intercept a significant amount of ground-water flow through hills and hillsides.

Lateral Distance from Mined Area

Among the 38 samples from wells less than 500 ft from mined areas, the 9 samples from wells more than 500 ft but less than 1,000 ft from mined areas, and the 11 samples from wells more than 1,000 ft from mined areas, concentrations of several mine-drainage constituents are higher than background. Sulfate, manganese, acidity, aluminum, zinc, and hardness concentrations are highest in samples collected at distances less than 500 ft downgradient of mined sites. At distances greater than 1,000 ft from mined sites, the concentrations typically are at or below background concentrations. In general, highest concentrations were detected within 1,000 ft from the mined sites (fig. 8). Sulfate and iron concentrations not only exceeded background near mined sites but also at distances more than 1,000 ft downgradient of the mined site. The relation between pH and distance is not as clear as for other constituents, but there is only one pH value below 6.0 at distance greater than 1,000 ft, compared to nine pH values at sites sampled less than 1,000 ft, and eight of the nine pH values less than 6.0 are at sites

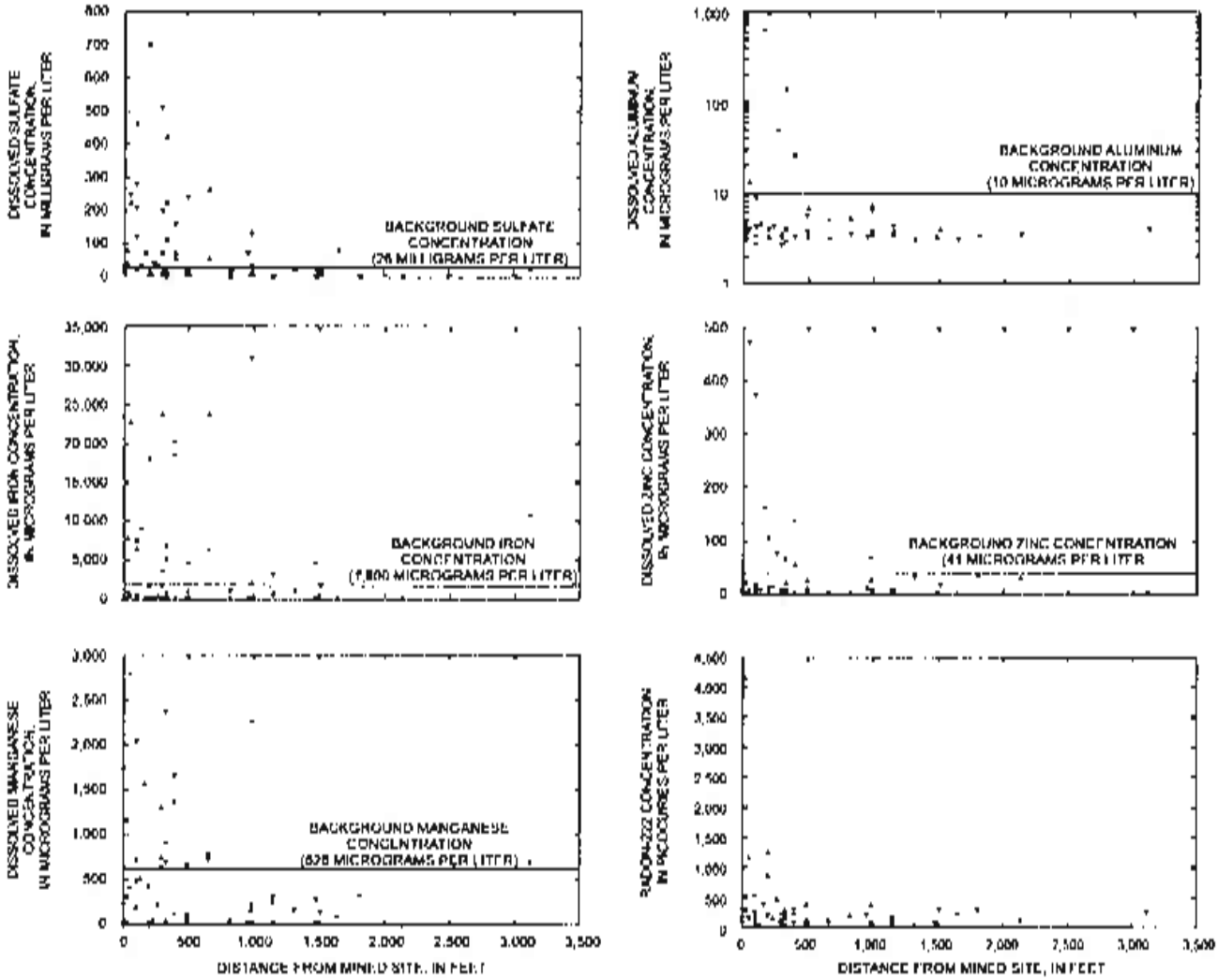


Figure 8. Concentrations of selected mine-drainage constituents downgradient of reclaimed surface coal mines.

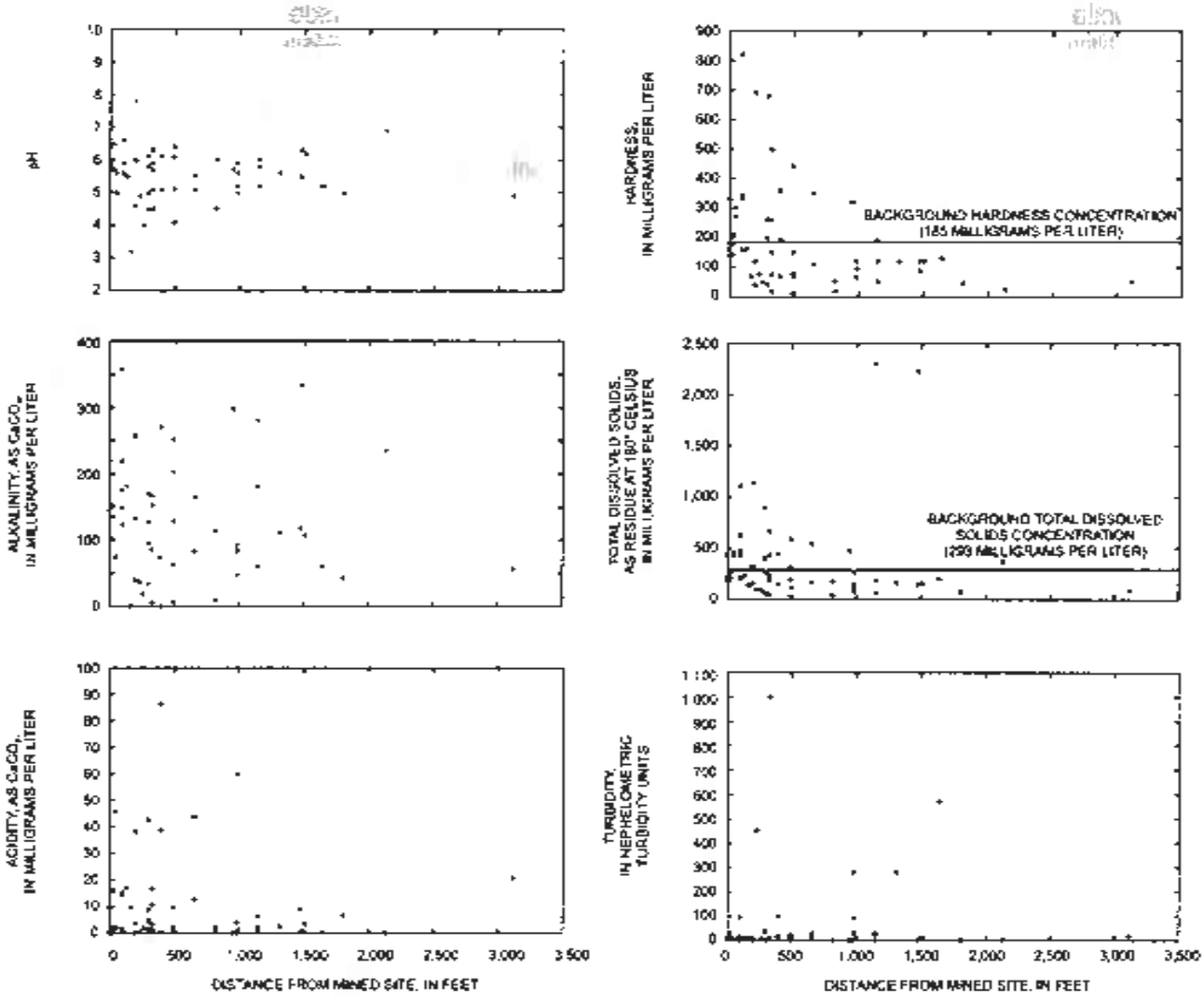


Figure 8. Concentrations of selected mine-drainage constituents downgradient of reclaimed surface coal mines.—Continued

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Vertical Extent and Relation to Depth of Wells

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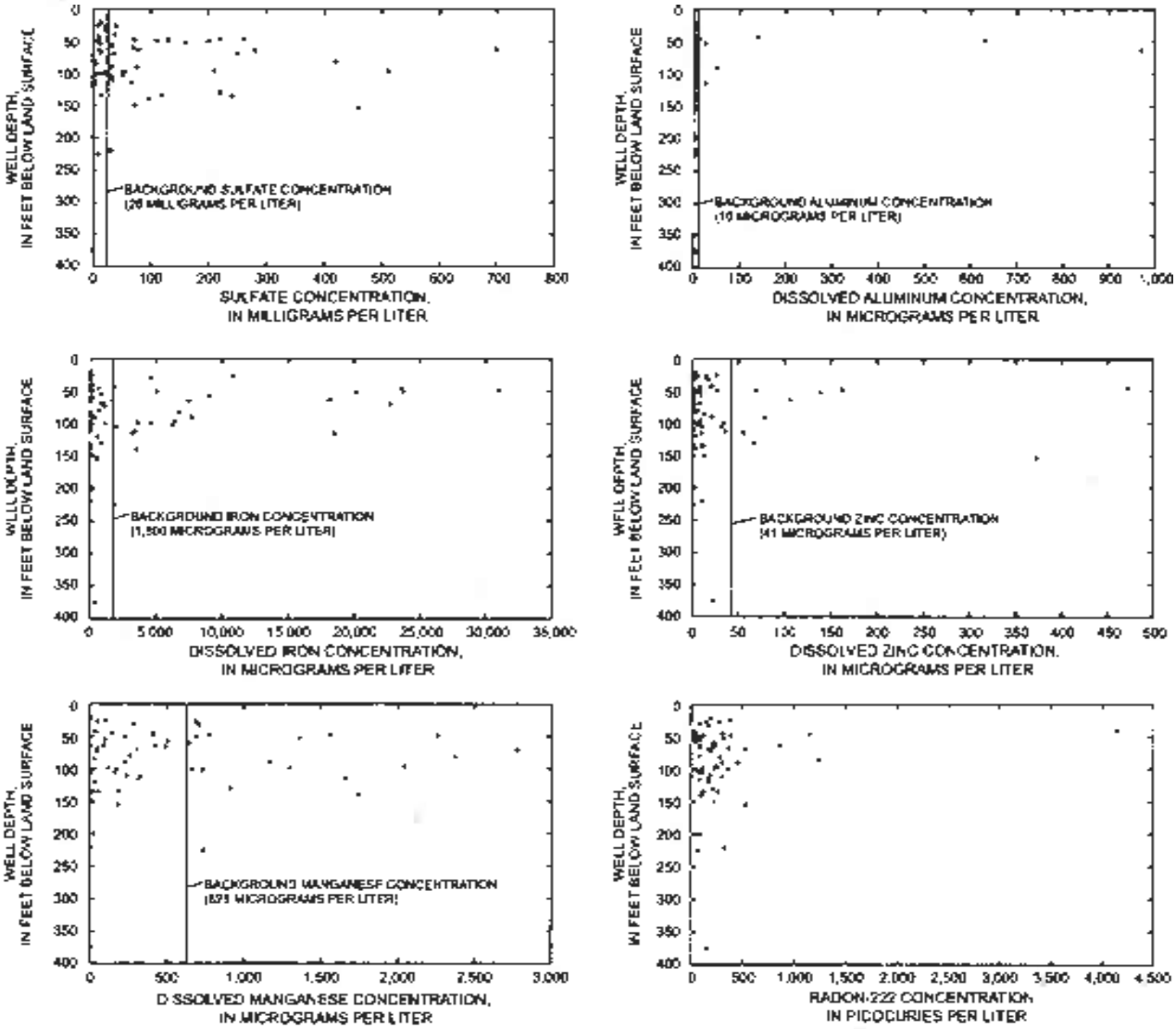

Figure 9. Vertical extent of mine-drainage constituents in ground water downgradient of reclaimed surface coal mines.

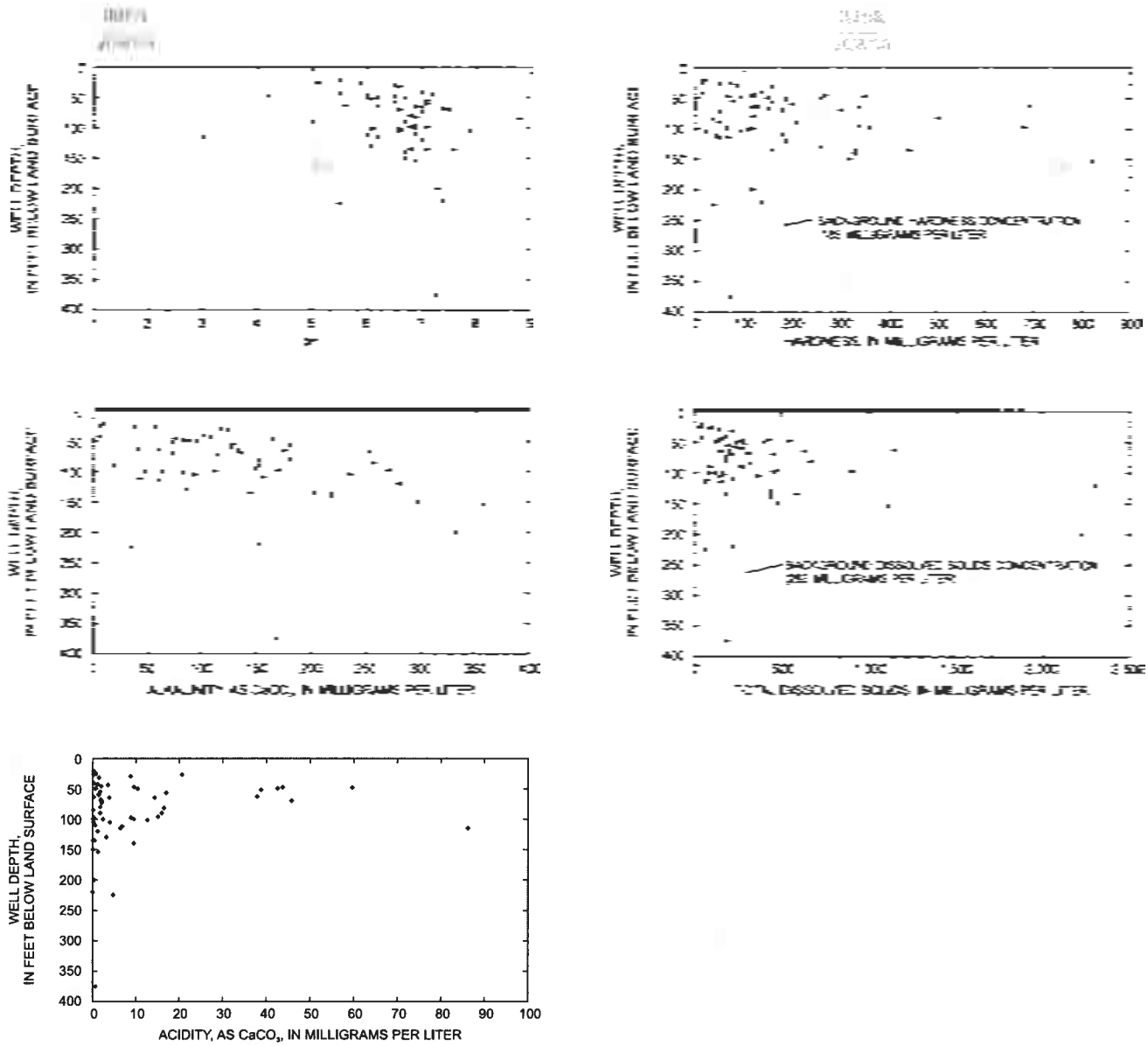

Figure 9. Vertical extent of mine-drainage constituents in ground water downgradient of reclaimed surface coal mines.—Continued

solids are greatest in wells in mined areas with well depths between about 50 and 150 ft (fig. 9). Concentrations of sulfate, iron, manganese, aluminum, and zinc are greater than background both in shallow wells (20 to 50 ft of depth) and in wells more than 50 to 150 ft deep. Acidity concentrations are also greatest at well depths less than 150 ft. Concentrations of zinc and radon are greatest at depths of less than 50 ft. Few concentrations of total dissolved solids and hardness exceed background at depths less than 50 ft and patterns on figure 9 indicate that these constituents' concentrations and alkalinity increase with increased well depth. The pH can be less than 6.0 at any depth, but several wells had less than 6.0 pH at depths less than 50 ft. In this study, the depth of the interception of the mine-drainage constituents is not known, only the well depth, but the mine-drainage constituents may flow through the aquifer at depths less than 50 ft according to other researchers who have shown that shallow wells and springs typically show higher concentrations of mine-drainage contaminants than deeper wells (McCurry and Rauch, 1986).

The increased concentrations of mine-drainage constituents from wells 20 to 150 ft deep compared to wells more than 150 ft deep likely coincide with the greater density of fractures available to transport mine-drainage constituents in the first 150 ft of strata. Fewer fractures at depths greater than 150 ft may result in less flow of ground water transporting mine-drainage constituents from the surface to greater depths. Of the 16

shallowest wells (less than 50 ft deep), 9 are in valleys, and water in these wells may have lower concentrations of mine-drainage constituents due to dilution of ground water as it intercepts and mixes with ground water from deeper regional or subregional flow paths.

Relation to Topographic Location

Topographic location has been reported to be a significant factor for both well yield and ground-water quality (McElroy, 1988). The effects of topographic location, such as hilltop, hillside, or valley locations, on ground-water quality downgradient from reclaimed surface mines are examined. Because only one downgradient well is on a hilltop, the hilltop and hillside locations were combined in evaluating water quality. Figure 10 shows box plots of sulfate, iron, and turbidity concentrations in unmined and mined areas in hilltop/hillside locations (referred to as hill locations) versus valley locations.

Results of multiple comparison tests using a multiple-step Kruskal-Wallis test (Helsel and Hirsch, 1992, p. 200-207; Ott, 1993) on ranks of the data indicate that very few mine-drainage contaminants in this study are shown to differ significantly based on topographic location. The tests indicate that turbidity was significantly greater at mined hill locations (median of 7.8 NTU) compared to unmined hill locations (median of 2.0 NTU). Sulfate concentrations were not significantly different among topographic locations. The median iron concentration of 2,000 µg/L at mined valley locations is significantly

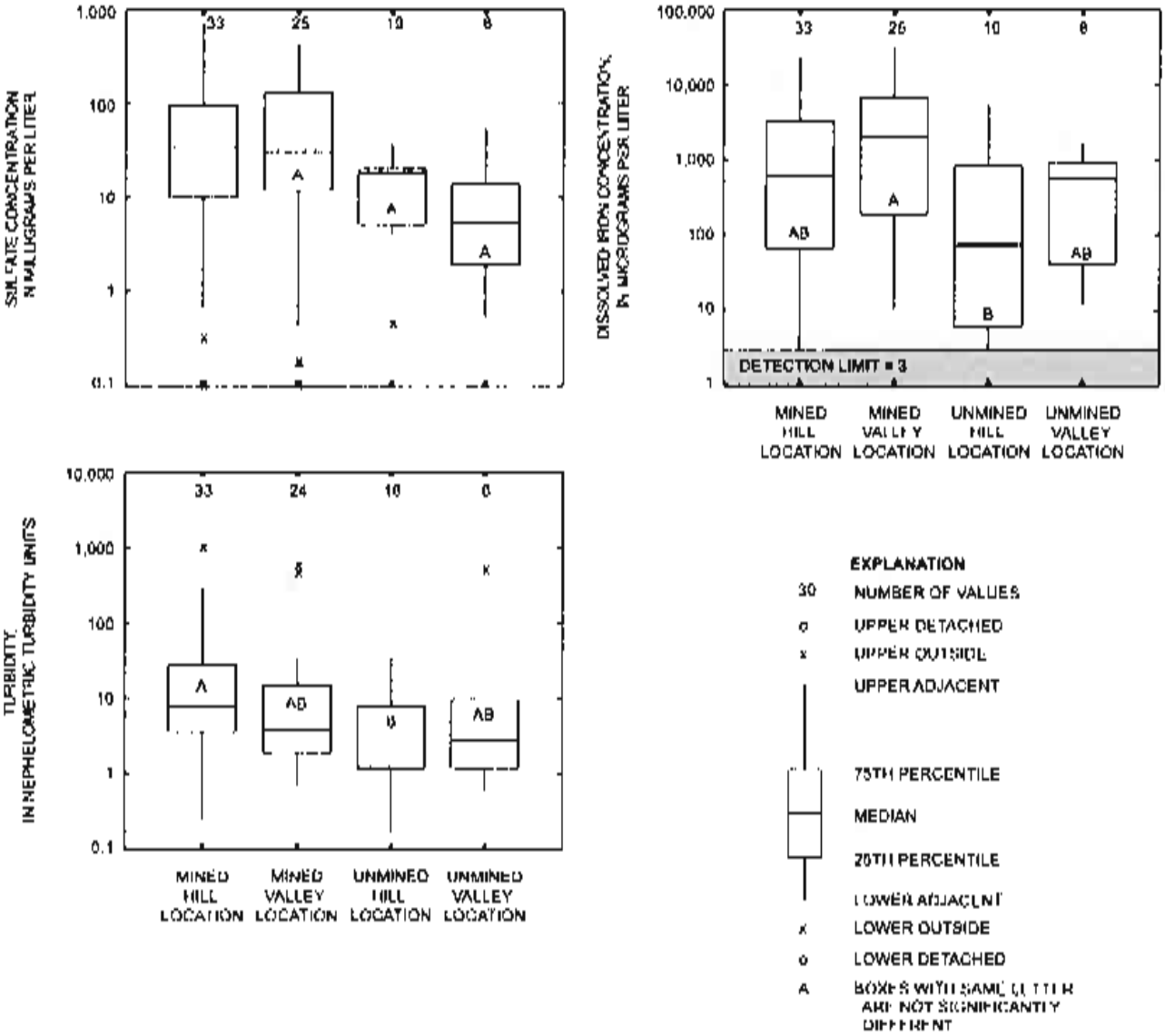

Figure 10. Distribution of constituent concentrations in ground-water samples from wells in hill and valley locations downgradient of reclaimed surface coal mines.

greater than the median iron concentrations of 74 µg/L at unmined hill locations. The median iron concentration for the mined hill location (590 µg/L) is greater, although not statistically significant, by nearly an order of magnitude than median iron concentration for the unmined hill location (74 µg/L).

The general trends (that are not statistically significant) relating topographic location to concentrations of mine-drainage constituents in well water are that the constituents downgradient of mined locations occur at higher concentrations than constituents in unmined areas, for both hill and valley locations. The downgradient hill locations have higher concentrations than the valley locations, except for iron. The higher concentrations of some mine drainage contaminants at hill locations is likely due to the fact that most mines are on hill slopes, and so the hill locations are closer to the mined areas.

Relation to Ground-Water Age

The residence time of water in the rock can affect concentrations of mine-drainage constituents. Ground-water age was measured in an attempt to estimate residence time in the rock. For samples collected from wells downgradient of reclaimed surface coal mines, 54 of 58 have age data (30 in the high-sulfur coal region and 24 in the low-sulfur coal region). Thirteen wells in unmined areas have age data for ground water. Ground-water age data presented within this report are characteristic of ground water from wells completed in fractured-bedrock aquifers and should not be assumed to be characteristic of ground-water flow in near-surface regolith or the unsaturated zone. The age data presented in this report represent the portion of ground water typically used by domestic homeowners and smaller public supplies as a source of drinking water and mixing of older and younger waters is likely. Obviously, younger water occurs in near-surface regolith, the unsaturated zone, and very shallow bedrock. Such shallow water (less than 50 ft in depth) typically has a chemical composition more characteristic of precipitation than of deeper ground water (Kozar, 1996). Methane and hydrogen sulfide can cause degradation or stripping of CFC compounds, especially for CFC-11 and CFC-113. Methane and hydrogen-sulfide compounds commonly are present in ground water near coal-bearing rock and near coal mines. Hydrogen sulfide was not analyzed in this study, but methane data are listed in appendix 1. The CFC compound data used to estimate ages are given in appendix 2 (G. Casile, U.S. Geological Survey, written commun., 2004).

Other researchers have indicated that because of increased residence time, specific conductance and dissolved solids increase with depth in hillside wells (Hawkins and others, 1996). Wells in valleys are reported to receive mixtures of various aged waters (Brady and others, 1996) and have greater yields (Clark and others, 1976).

A Spearman's rank correlation was conducted to compare the age data to the significant water-quality data, specifically the mine-drainage constituents listed previously. Spearman's rank correlation was also used to compare the age data to the distance data by topographic location (hill and valley). No significant correlations were shown between any mine-drainage constituent and age. The only mine-drainage constituents correlative with age (indicated by a correlation coefficient greater than 0.50) were pH and alkalinity, and then only in the low-sulfur coal region, and acidity, only in the high-sulfur region, where correlation coefficients were 0.55, 0.52, and 0.53, respectively, indicating an increase with increased age, but the relation is not statistically significant (fig. 11). No significant

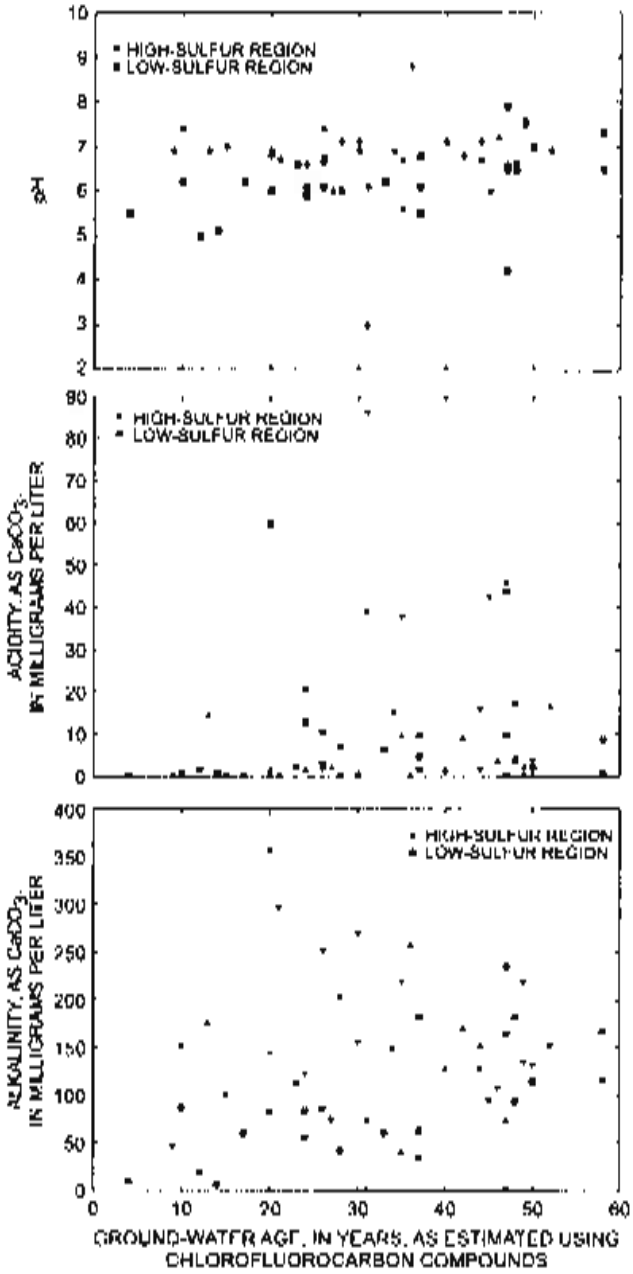

Figure 11. Comparison of selected mine-drainage constituents and ground-water age downgradient of reclaimed surface coal mines in high-sulfur and low-sulfur coal regions.

correlation exists between the distance data and the age data for well waters downgradient from mines, by topographic-location group (hill and valley) (fig. 12). However, at distances greater than 1,000 ft from the mined area, age of water from wells on hills shows an increase with increased distance from the mine (fig. 12). At other distances, any age of well water between 4 and 58 years is reported for both hill and valley topographic locations (fig. 12).

The lack of significant correlation among mine-drainage constituents, age, distance from mined areas, and topographic location may be the result of several factors. These factors include mixing of ground-water ages in water intercepted by the wells, hydraulic impacts of mining (disturbance of rock from blasting), configuration of land surface (slope and relief), and rock orientation relative to mined areas and wells sampled.

Several conditions may result in wells in mined areas intercepting ground water of mixed ages (or residence times): (1) strong indications of mixed residence times were seen in 10 of 54 wells (19 percent) for which CFC data were used to estimate ground-water residence time as shown in appendix 2 and figure 12 (G. Casile, U.S. Geological Survey, written commun., 2004); (2) some samples have old ages (46->58 years) determined from CFC data but are very near (35 ft or less) to the mined area (fig. 12), which would require an extremely small hydraulic conductivity to move the ground water only a short distance over decades (assuming that the hydraulic gradient is 0.1 or greater), and that is unlikely; and (3) the wells are open hole completions into the fractured bedrock and the fractures may supply ground water of differing residence times based on fracture lengths, fracture depths, and interconnection of multiple fractures of variable lengths and depths.

A significant hydraulic impact from mining is rock disturbance, such as caused by blasting at the mine site. Blasting can cause increased fracture density and open more interconnected pathways for a greater volume of mine-drainage constituents to flow downgradient of the mined site, perhaps at a more rapid rate than that prior to the disturbance from mining. The effects

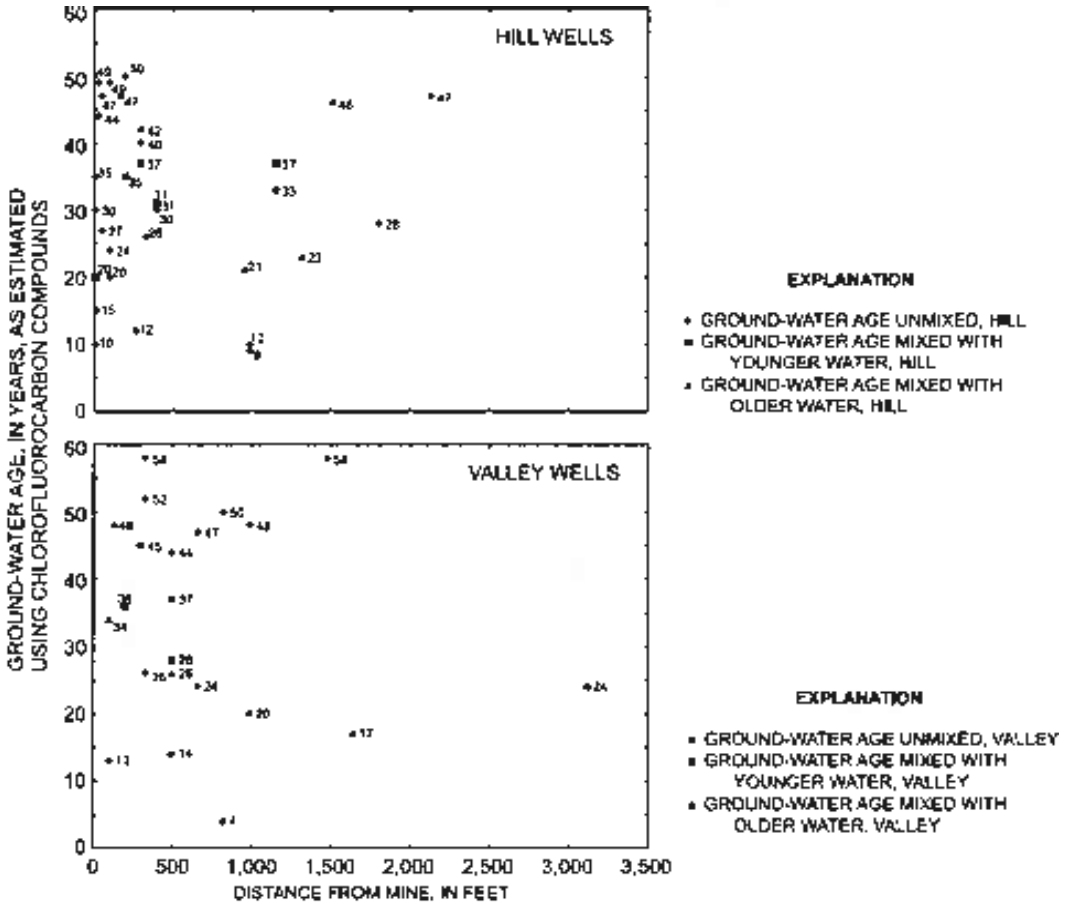

Figure 12. Age of well water at various distances and hill and valley topographic locations downgradient of reclaimed surface coal mines. (Samples listed with mixed ages are estimated by interpretation of chlorofluorocarbon data by G. Casile, U.S. Geological Survey, written commun., 2004).

would vary from site to site dependent on amount of overburden removal and blasting done.

Configuration of land surface and rock orientation and the variability of these factors within the study area may be related to ground-water residence times. For example, wells in hilltop settings in mined areas with steep slopes and terrain may have "younger" aged ground water and higher concentrations of mine-drainage constituents than wells in mined areas with less steep slopes and terrain. Orientation of bedrock with downgradient directions of ground-water flow could result in younger ground-water ages, but orientation (downdip, updip, or along strike relative to the location of the mine) did not correlate with high concentrations of mine-drainage constituents in this study. Also, mixing of older and younger ground water is more likely in valley settings than in hilltop or hillside settings.

A comparison of mine-drainage constituents in "young" ground water (age of water less than or equal to time elapsed since the reclamation was completed) to representative mine-drainage constituent concentrations in unmined area ground-water samples could not be conducted because the age of most ground waters sampled exceeded the time elapsed since reclamation completion. Twelve years is the maximum elapsed time since reclamation completion in the sample population. Yet, of all samples with ages exceeding 12 years, most samples had at least one mine-drainage constituent concentration exceeding background levels.

The median apparent age of ground water from 24 wells (table 6) sampled in mined areas of the low-sulfur coal region was 30 years. Samples from four wells in unmined areas of the low-sulfur coal region indicated a similar median apparent ground-water age of 25 years. No statistically significant difference in apparent ground-water age could be determined between mined and unmined areas in the low-sulfur coal region. In the high-sulfur coal region, the median apparent age of ground water from 9 wells in unmined areas was 44 years and the median apparent age of ground water from 30 wells in mined areas was 33 years. Although the median apparent ground-water age for unmined areas in the high-sulfur coal region was approximately 11 years older than mined areas, no statistical difference in the age of mined and unmined areas could be established.

Table 6. Statistical summaries of ground-water age comparing mined and unmined areas and topographic location of wells in the high-sulfur and low-sulfur coal regions.

[>, greater than]

Coal region	Sample size	Ground-water age, in years		
		Median	Maximum	Minimum
High-sulfur coal region				
Mined areas	30	33	52	9
Unmined areas	9	44	50	10
Low-sulfur coal region				
Mined areas	24	30	>58	4
Unmined areas	4	25	27	11
High-sulfur coal region (All wells sampled)				
Hilltop wells	3	44	50	35
Hillside wells	26	31	50	9
Valley wells	10	35	52	10
Low-sulfur coal region (All wells sampled)				
Hilltop wells	2	18	25	11
Hillside wells	11	28	47	10
Valley wells	15	26	>58	4
High-sulfur coal region (Mined areas only)				
Hilltop wells	1	35	35	35
Hillside wells	21	31	50	9
Hilltop and hillside wells	22	31	50	9
Valley wells	8	34	52	13
Low-sulfur coal region (Mined areas only)				
Hilltop and hillside wells	10	30	47	10
Valley wells	14	32	58	4

Ground water in mined areas in the high-sulfur coal region is typically older and has less variability in age than that in the low-sulfur coal region (table 6 and appendix 2). Major factors likely responsible for differences in ground-water age between the two coal regions are topographic relief and slope. Typical basin slopes are 60 percent greater and relief is about 20 percent greater in the low-sulfur coal region than in the high-sulfur coal region, which may result in less residence time, or younger ground-water age in general, in the low-sulfur coal region. Topographic location was the only factor that has a discernible pattern (though not statistically significant) with apparent ground-water age, and only in the low-sulfur coal region (table 6). Within the low-sulfur coal region, ground water from wells on hilltops (median age of 18 years) was younger than water from wells in valley settings (median age of 26 years). Ground-water age data indicate ground-water flow from ridges or hillsides to streams or subregional aquifers in valley bottoms. Within the high-sulfur coal region, median age of ground water for hilltop, hillside, and valley wells was 44, 31, and 35 years, respectively.

Although data from this study do not show a statistically significant relation between topographic setting and ground-water age, in a previous investigation, 25 wells were sampled randomly within the low-sulfur central coal region as part of an areal assessment of ground-water age within the Appalachian Plateaus Physiographic Province in the Kanawha-New River Basin. The study did not specifically sample wells downgradient from reclaimed surface mines; the wells selected for sampling were randomly chosen. For these 25 wells, the median age of ground water was estimated to be 30 years (Kozar, 1998), and median ground-water age for wells sampled in hilltop, hillside, and valley settings were 13, 29, and 42 years, respectively (Kozar, 1998). This finding is similar to age estimates determined as part of this investigation where well sampling yielded median age estimates of 18, 28, and 26 years for wells sampled within the low-sulfur coal province from hilltop, hillside, and valley settings, respectively (table 6). The major difference being that ground water in valley settings in primarily mined areas (median age of 26 years) appears to be significantly younger than that documented in valley settings in the earlier investigation (median age of 42 years) where wells were selected randomly. Younger waters in valley settings in mined areas may indicate a faster ground-water transit time in valley settings because of the enhanced fracturing of rock that is common as a result of both surface and underground coal mining.

Summary and Conclusions

Surface coal mines have historically had major impacts on the quality of shallow ground water. Since about 1977, changes in mining practice have been mandated to result in reclamation of surface-mined areas. Although current programs in Pennsylvania and West Virginia monitor water quality in and near surface coal mines, this study examines whether post-reclamation ground-water quality near coal mines is similar to ground-water quality observed in unmined areas in the bituminous coal fields of the northern Appalachian coal region (high-sulfur coal region) and the central Appalachian coal region (low-sulfur coal region). This study also examines the spatial and temporal extent of chemical constituents derived from mine drainage in ground water downgradient of reclaimed surface-coal-mined areas. This effort is part of the USGS NAWQA Program assessment of factors that affect ground-water quality, such as mining land use.

Samples collected from 58 wells (30 in the high-sulfur coal region and 28 in the low-sulfur coal region) about 2,500 ft or less downgradient of surface coal mines that were reclaimed during 1985-96 were compared to 25 samples from wells (20 in the high-sulfur and 5 in the low-sulfur coal region) in unmined areas. Samples from mined areas also were compared to background concentrations (estimated from the 90th percentile of concentrations in unmined areas) of selected constituents (sulfate, iron, manganese, aluminum, zinc, hardness, and dissolved solids). The wells were all private domestic supply wells of depths generally less than 250 ft. Ground water in these wells flows into open-hole sections in the gently dipping, relatively flat-lying fractured units of Pennsylvanian age, which may include sandstone, siltstone, shale, limestone, and coal.

Constituents examined include common inorganics (particularly constituents related to mine drainage such as sulfate, iron, manganese, aluminum, pH, acidity, alkalinity, hardness, and total dissolved solids), various trace elements (radon, lead, nickel, and zinc), and VOCs. Ground-water age was calculated from CFC compounds.

The mine-drainage constituents in ground water downgradient of the mined areas result from the transport of pyrite-oxidation products from the mined site and from neutralization reactions by the dissolution of calcareous materials introduced at the mine site or along the ground-water-flow path. The disturbance of the rock during blasting, drilling, or excavating causes fracturing of rocks and exposes rocks to additional recharge, mechanical weathering, and geochemical reactions.

Ground water downgradient of surface coal mines has significantly higher concentrations of sulfate, iron, manganese, aluminum, pH, hardness (calcium and magnesium), turbidity, and specific conductance compared to ground water in unmined areas. In addition, concentrations of zinc and detections of BTEX compounds (in the high-sulfur region only) were greater downgradient of surface coal mines compared to unmined areas. High concentrations of iron and manganese were ubiquitous throughout the study area. Sixty-four percent of samples downgradient of surface mines and 36 percent of samples from unmined areas exceeded the USEPA SMCL of 300 µg/L for iron, and 71 percent of samples downgradient from surface mines and 56 percent of samples from unmined areas exceeded the USEPA SMCL of 50 µg/L for manganese.

Sulfate, hardness, calcium, and magnesium concentrations were significantly greater in the high-sulfur coal region downgradient of reclaimed surface coal mines compared to downgradient of reclaimed surface coal mines in the low-sulfur coal

region and unmined locations in either region. Specific conductance, pH, and radon downgradient of reclaimed surface coal mines were significantly greater in samples from the high-sulfur region compared to samples from the low-sulfur region sites.

Concentrations of several mine-drainage constituents (sulfate, manganese, aluminum, zinc, and hardness) were greatest and generally exceeded background less than 500 ft downgradient of surface-coal-mine sites but decreased to less than background concentrations at distances more than 1,000 ft downgradient. Concentrations of sulfate, hardness, and total dissolved solids were greatest in wells with depths between 50 and 150 ft but generally were less than background concentrations in wells deeper than 150 ft. Iron, manganese, and aluminum concentrations exceeded background in shallow wells (less than 50 ft deep) and in wells 50 to 150 ft deep. Alkalinity concentrations and pH were variable at all well depths. Radon and zinc concentrations were highest in shallow wells compared to wells more than 50 ft deep. Although turbidity levels were significantly greater at hill locations downgradient from surface-coal-mined sites compared to hill locations in unmined areas, mine-drainage constituents did not vary significantly by topographic location (hill as compared to valley).

The median age of ground water sampled in the low-sulfur coal region in mined and unmined areas was 30 and 25 years, respectively. The median ground-water age sampled in the high-sulfur coal region in mined and unmined areas was 33 and 44 years, respectively.

In mined areas, ground water in the high-sulfur coal region typically was older than ground water in the low-sulfur coal region, though not significantly. Ground-water age in mined areas typically was greater than time elapsed since mining completion and did not significantly correlate with concentrations of any mine-drainage constituent.

In comparing ground-water age to distance from mined areas and to topographic location, no significant correlations were found but a few observations are noteworthy. In mined areas, median ground-water ages increased from hill to valley locations for wells sampled in the low-sulfur and high-sulfur coal regions. In samples from wells in hill locations more than 1,000 ft from a mined area, a pattern was observed that ground-water age increased with increased distance from the mined area.

The lack of significant correlation among mine-drainage constituents, age, distance from mined areas, and topographic location may be the result of several factors. These factors include mixing of ground-water ages in water intercepted by the wells, hydraulic impacts of mining (disturbance of rock from blasting), configuration of land surface (slope and relief), and rock orientation (dip and strike) relative to mined areas and wells sampled. The mixing is likely related to the open-hole completion wells that intersect fractures of varying numbers, depths, lengths, and are interconnected with other fractures. In addition, the effects of blasting vary with the amount of overburden removed, and slope, relief, and structural orientation vary across the study area.

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Appendixes 1-3

Appendix 1a—Dissolved gas and recharge temperature data for 24 wells sampled downgradient from reclaimed surface coal mines in the low-sulfur coal region

Appendix 1b—Dissolved gas and recharge temperature data for 9 wells sampled downgradient from reclaimed surface coal mines in the high-sulfur coal region

Appendix 1c—Dissolved gas and recharge temperature data for 5 wells sampled in unmined areas in the low-sulfur coal region

Appendix 1d—Dissolved gas and recharge temperature data for 8 wells sampled in unmined areas in the high-sulfur coal region

Appendix 2a—Chlorofluorocarbon data and apparent ground-water-recharge dates for 24 wells sampled in the low-sulfur coal region downgradient from reclaimed surface coal mines

Appendix 2b—Chlorofluorocarbon data and apparent ground-water-recharge dates for 30 wells sampled in the high-sulfur coal region downgradient from reclaimed surface coal mines

Appendix 2c—Chlorofluorocarbon data and apparent ground-water-recharge dates for 5 wells sampled in unmined areas of the low-sulfur coal region

Appendix 2d—Chlorofluorocarbon data and apparent ground-water-recharge dates for 9 wells sampled in unmined areas in the high-sulfur coal region

Appendix 3—Site identification numbers, local well names, coal-sulfur-content region, land-use classification, topographic location, distance from mine, and well depth for wells sampled

Appendix 1a. Dissolved gas and recharge-temperature data for 24 wells sampled downgradient from reclaimed surface coal mines in the low-sulfur coal region.

[°C, degrees Celsius; mg/L, milligrams per liter; *, possible methane degradation of CFCs; ND, no data available]

Local identifier	Field water temperature (°C)	N_2 (mg/L)	Ar (mg/L)	O_2 (mg/L)	CO_2 (mg/L)	CH_4 (mg/L)	Recharge elevation	Recharge temperature (°C)	Excess air (mg/L)
Ral 0211	14.9	15.848	0.5764	0.07	76.73	0.0000	2,584	14.7	0.8
	14.9	16.013	.5760	.06	78.32	.0000	2,584	15.2	1.1
Fay-0274	15.0	16.456	.6142	6.99	65.56	.0000	2,000	12.0	.3
	15.0	16.450	.6146	7.39	62.86	.0000	2,000	12.0	.2
Fay-0275	11.5	17.398	.6527	3.86	16.85	.0000	2,440	8.8	.3
	11.5	17.594	.6532	4.03	16.79	.0000	2,440	9.1	.6
Fay-0276	13.4	17.975	.6384	.69	70.41	.0000	2,400	11.6	1.9
	13.4	18.220	.6422	.48	70.45	.0000	2,400	11.7	2.2
Fay-0279	11.3	28.490	.8502	.20	46.87	.0072	2,240	8.2	11.0
	11.3	19.645	.6845	.57	37.84	.8104	2,240	9.8	2.8
Grb-0284	12.3	26.485	.8180	1.13	46.46	.0017	3,320	6.4	9.0
Fay 0272	12.6	19.406	.6869	.07	78.42	.0021	2,480	8.7	2.3
	12.6	19.334	.6848	.08	76.83	.0028	2,480	8.8	2.3
Boo-0260	13.2	20.728	.7528	.00	14.69	*7.8175	1,760	5.3	1.8
	13.2	20.860	.7589	.00	14.99	*7.6909	1,760	4.9	1.7
Boo-0258	13.6	18.446	.6697	.61	37.97	.0000	1,640	10.0	1.3
	13.6	18.132	.6618	1.34	37.52	.0000	1,640	10.3	1.1
Fay 0270	13.7	21.286	.6966	.10	50.45	.0402	2,000	12.0	5.1
	13.7	21.745	.7095	.09	50.66	.0234	2,000	11.4	5.3
Boo-0259	14.2	24.680	.8839	.00	5.73	*8.4649	1,560	ND	3.0
	14.2	24.689	.8904	.00	5.82	*8.0106	1,560	ND	2.7
Boo-0256	13.4	19.963	.6870	.23	73.32	.2458	1,160	11.8	3.2
	13.4	20.409	.6944	.27	82.90	.3121	1,160	11.8	3.6
Boo-0257	15.0	7.361	.2674	.00	102.81	*13.3136	1,180	ND	-1.3
	15.0	8.546	.3149	.00	123.63	*11.0795	1,180	ND	-1.2
Kan-0935	13.8	20.369	.6990	.08	24.48	.0042	1,440	10.7	3.4
Nic-0212	13.0	19.171	.6865	2.04	51.92	.0000	1,680	9.5	1.9
	13.0	19.168	.6896	1.64	52.68	.0000	1,680	9.1	1.7

Appendix 1a. Dissolved gas and recharge-temperature data for 24 wells sampled downgradient from reclaimed surface coal mines in the low-sulfur coal region.—Continued

(°C, degrees Celsius; mg/L, milligrams per liter; *, possible methane degradation of CFCs; ND, no data available)

Local identifier	Field water temperature (°C)	N_2 (mg/L)	Ar (mg/L)	O_2 (mg/L)	CO_2 (mg/L)	CH_4 (mg/L)	Recharge elevation	Recharge temperature (°C)	Excess air (mg/L)
Nic-0209	11.8	21.143	0.7334	0.13	24.83	0.0713	1,880	7.8	3.3
	11.8	21.169	.7385	.11	28.64	.0721	1,880	7.3	3.1
Nic-0213	13.8	20.882	.7045	.20	75.18	.2022	2,200	10.0	4.1
	13.8	29.378	.8527	.23	80.83	.1896	2,200	9.6	12.5
Kan-0938	14.4	19.398	.6751	.29	17.24	.3522	1,580	11.4	2.7
	14.4	19.507	.6780	.30	16.08	.3997	1,580	11.3	2.8
Nic-0214	12.8	20.219	.6638	.31	83.38	.3837	2,440	13.2	4.7
	12.8	19.857	.6576	.30	82.85	.3826	2,440	13.2	4.3
Nic-0215	12.4	22.711	.7994	.00	11.61	*6.3042	2,440	3.2	3.2
	12.4	25.044	.8377	.00	11.61	*6.1873	2,440	3.3	5.6
Nic-0211	12.5	16.233	.6225	.00	36.32	*5.2565	2,480	9.9	-.4
	12.5	15.919	.6182	.00	38.59	*5.5752	2,480	9.7	-.8
Nic-0219	12.5	21.044	.7220	.22	59.44	.2434	2,800	7.5	3.7
	12.5	20.500	.7134	.23	58.51	.2621	2,800	7.4	3.1
Nic-0217	12.5	23.409	.7518	.60	59.81	.6122	2,520	9.0	6.5
Nic-0218	12.1	18.126	.6303	3.12	198.76	.0000	2,560	12.8	2.5

Appendix 1b. Dissolved gas and recharge temperature data for 9 wells sampled downgradient from reclaimed surface coal mines in the high-sulfur coal region.

[°C, degrees Celsius; mg/L, milligrams per liter; *, possible methane degradation of CFCs]

Local identifier	Field water temperature (°C)	N_2 (mg/L)	Ar (mg/L)	O_2 (mg/L)	CO_2 (mg/L)	CH_4 (mg/L)	Recharge elevation	Recharge temperature (°C)	Excess air (mg/L)
LE411	16.7	20.504	0.7017	0.05	0.92	*16.1643	1,070	11.3	3.5
WS973	14.5	21.462	.7436	.29	39.05	.0047	1,090	8.4	3.4
	14.5	27.119	.8468	2.00	39.35	.0042	1,090	7.7	8.7
MN612	14.1	21.350	.6608	.12	142.84	.0000	1,140	18.4	6.6
	14.1	21.496	.6628	.15	138.87	.0004	1,140	18.5	6.8
BA012	14.4	20.968	.7174	.11	81.45	.5526	1,700	9.4	3.7
CR726	13.2	21.725	.7497	.04	59.04	.0283	1,270	8.0	3.6
	13.2	22.557	.7642	.07	61.79	.0283	1,270	8.0	4.4
AR106	12.8	13.095	.4944	.00	32.26	*25.5844	980	22.8	-.6
	12.8	14.473	.5417	.03	32.77	*25.7054	980	19.0	-.2
SO853	11.8	20.217	.7103	.02	116.01	.0624	2,380	7.8	2.7
	11.8	20.510	.7224	.01	115.41	.0663	2,380	7.1	2.7
JE423	11.9	23.231	.7974	.06	61.55	.0090	1,460	5.5	4.2
	11.9	22.854	.7897	.14	60.91	.0093	1,460	5.7	3.8
JE424	11.6	23.479	.7240	.07	257.99	.0533	1,670	13.8	7.7

Appendix F6. Dissolved gas and recharge temperature data for 5 wells sampled in unmined areas in the low-sulfur coal region.

[°C, degrees Celsius; mg/L, milligrams per liter; *, possible nitrate degassing; cfc/HCs, DVT sample could not be analyzed for organic compounds due to extensive interference effects]

Local identifier	Field water temperature (°C)	N_2 (mg/L)	Ar (mg/L)	O_2 (mg/L)	CO_2 (mg/L)	CH_4 (mg/L)	Recharge elevation	Recharge temperature (°C)	Excess air (mg/L)
Rul-0025	11.2	17.881	0.6525	6.10	34.06	0.0000	2,550	9.1	7.19
Ken-2927	13.3	21.625	.7217	.04	47.59	.0000	1,000	9.4	3.93
Cas-0141	12.5	18.271	.6255	.07	23.51	*6.7144	3,400	17.2	-.59
FA509	13.2	19.513	.6635	4.91	48.54	.0000	2,385	10.1	.3
	13.2	19.525	.6648	5.18	47.45	.0000	2,555	10.3	.6
LE412	19.3	13.598	.5085	.287	12.44	*8.7833	1,100	21.7	-.3
	19.3	13.344	.4987	.187	12.43	*8.5665	1,100	22.6	-.4

Appendix 1d. Dissolved gas and recharge temperature data for 8 wells sampled in unmined areas in the high-sulfur coal region.

[°C, degrees Celsius; mg/L, milligrams per liter; *, possible methane degradation of CFCs]

Local identifier	Field water temperature (°C)	N_2 (mg/L)	Ar (mg/L)	O_2 (mg/L)	CO_2 (mg/L)	CH_4 (mg/L)	Recharge elevation	Recharge temperature (°C)	Excess air (mg/L)
Nic-0208	12.5	20.953	0.7389	0.03	32.31	*1.9742	3,000	5.3	2.83
SO124	14.7	20.465	.7068	.06	2.37	*1.0421	700	11.2	3.25
SO126	13.4	21.988	.7236	9.03	51.25	.0003	1,120	11.6	5.09
	13.4	22.294	.7337	9.20	51.24	.0000	1,120	11.0	5.19
IN915	13.5	17.732	.6434	3.61	17.75	.0000	1,564	11.8	1.19
Bro-0269	13.4	21.553	.7383	.15	23.12	.0022	1,400	8.7	3.76
CR727	15.1	22.306	.7496	.15	31.93	.1026	1,485	8.7	4.6
	15.1	22.061	.7452	.15	26.63	.1097	1,485	8.7	4.3
AR108	12.8	26.568	.8278	.09	11.58	.0000	1,240	8.5	8.6
	12.8	26.311	.8208	.09	11.29	.0000	1,240	8.8	8.4
WE315	12.0	19.857	.7103	.10	41.44	.0000	1,120	9.0	2.0
	12.0	19.712	.7073	.65	42.17	.0000	1,120	9.1	1.9

Appendix 2a. Chlorofluorocarbon data and apparent ground-water-recharge dates for 24 wells sampled in the low-sulfur coal region downgradient from reclaimed surface coal mines.

[ft, feet; pg/kg, picograms per kilogram; pptv, parts per trillion per volume; <, less than date or age; >, greater than date or age. CFC sample collected for well Bso-0257, however, no data tabulated here due to sample degredation and inconclusive results.]

Local county identifier	Date	Time	Well depth (ft below land surface)	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
Sum-0105	06181998	1200	115	34.1	45.7	0.0	13.6	78.9	0.0	11.0	1965	33
				12.7	42.5	.0	5.1	73.5	.0	11.0		
				11.2	58.9	2.6	4.5	101.8	2.5	11.0		
Ral-0221	05271998	900	49.5	479.3	227.9	60.1	236.4	479.2	72.1	14.7	1988	10
				442.6	216.8	57.1	218.3	455.9	68.5	14.7		
				467.9	225.5	59.4	230.8	474.0	71.3	14.7		
Fay-0274	05211998	1100	47.5	489.4	304.4	209.4	205.1	552.0	210.5	12.0	1951	47
				490.7	299.1	219.3	205.6	542.3	220.4	12.0		
				493.6	293.4	227.6	206.8	531.9	228.7	12.0		
Fay-0275	05121998	1100	21.25	684.3	324.1	101.9	247.0	515.2	86.8	9.0	1994	4
				686.2	334.0	96.1	247.7	530.9	81.9	9.0		
				686.9	325.4	101.4	248.0	517.1	86.4	9.0		
Fay-0276	05081998	1300	102	209.6	122.7	122.2	87.7	222.6	122.4	11.7	1974	24
				190.5	118.0	129.5	79.7	214.1	129.8	11.7		
				193.4	112.8	114.3	80.9	204.7	114.6	11.7		
Fay-0279	05261998	1400	30	.0	.0	.0	.0	.0	.0	9.0	<1940	>58
				.0	.0	.0	.0	.0	.0	9.0		
				2.5	1.8	.0	.9	2.9	.0	9.0		
Gab-0284	06161998	1100	112	14,684.4	110.3	30.0	4,708.5	158.2	22.4	6.4	1970	28
				2,104.9	53.1	12.9	674.9	76.2	9.6	6.4		
				1,888.1	51.5	13.0	605.4	73.9	9.7	6.4		

Appendix 2a. Chlorofluorocarbon data and apparent ground-water-recharge dates for 24 wells sampled in the low-sulfur coal region downgradient from reclaimed surface coal mines.—Continued

[ft, feet; pg/kg, picograms per kilogram; pptv, parts per trillion per volume; <, less than date or age; >, greater than date or age. CFC sample collected for well Boo-0257, however, no data calculated here due to sample degradation and inconclusive results.]

Local county identifier	Date	Time	Well depth (ft below land surface)	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date[2]	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
Fay-0272	05131998	1200	225	39.1	37.4	10.4	13.9	58.6	8.7	8.7	1951 with younger water	37
				35.2	32.3	7.0	12.5	50.6	5.9	8.7		
				37.0	36.1	8.2	13.1	56.6	6.9	8.7		
Boo-0260	05151998	1000	73	1.4	3.8	.0	.4	4.8	.0	5.1	1948	50
				.9	1.7	.0	.3	2.2	.0	5.1		
				.0	.0	.0	.0	.0	.0	5.1		
Boo-0258	05061998	1000	64	333.9	176.5	61.2	124.4	287.7	54.2	10.1	1981	17
				354.8	193.0	67.9	132.2	314.6	60.1	10.1		
				352.4	188.8	63.4	131.3	307.9	56.1	10.1		
Fay-0270	05221998	1200	27	4.1	109.3	4.8	1.7	195.3	4.8	11.7	1974	24
				154.1	266.7	17.2	63.5	476.6	17.0	11.7		
				4.9	114.4	5.1	2.0	204.4	5.1	11.7		
Boo-0259	06231998	1000	105	3.6	2.9	.0	1.4	5.0	.0	12.0	1951	47
				6.2	4.4	0	2.4	7.5	.0	12.0		
				11.9	11.4	0	4.7	19.4	.0	12.0		
Boo-0256	05041998	1300	130	76.2	96.4	13.8	30.6	167.8	13.3	11.8	1972	26
				69.7	101.0	15.5	28.0	175.8	14.9	11.8		
				67.9	102.2	13.8	27.3	177.9	13.3	11.8		

Appendix 2a. Chlorofluorocarbon data and apparent ground-water-recharge dates for 24 wells sampled in the low-sulfur coal region downgradient from reclaimed surface coal mines.—Continued

[ft, feet; pg/kg, picograms per kilogram; pptv, parts per trillion per volume; <, less than date or age; >, greater than date or age. CFC sample collected for well Boo-0257; however, no data tabulated here due to sample degradation and inconclusive results.]

Local county identifier	Date	Time	Well depth (ft below land surface)	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
Kac-0935	05051998	1100	100	130.9	86.0	13.4	50.1	143.4	12.2	10.7	1975 with older water	23
				169.2	108.5	22.2	64.7	180.9	20.2	10.7		
				183.8	116.0	20.6	70.3	193.4	18.7	10.7		
Nic-0212	05191998	1400	25.5	268.2	194.2	74.1	95.7	304.6	62.5	9.3	1984	14
				249.3	191.0	74.3	89.0	299.7	62.7	9.3		
				6,581.6	921.3	68.5	2,349.2	1,448.0	57.8	9.3		
Nic-0209	07081998	900	50	4.2	95.2	.0	1.4	137.9	0	7.6	1972	26
				3.9	122.5	.0	1.3	177.4	.0	7.6		
				2.6	120.2	.0	.8	174.1	.0	7.6		
Nic-0213	05281998	1400	48.38	122.6	186.3	.0	45.9	305.5	0	9.8	1978	20
				12.1	149.2	.0	4.5	244.7	0	9.8		
				3.2	175.5	.0	1.2	287.8	0	9.8		
Kac-0938	06221998	1000	105	2.7	.8	.0	1.1	1.4	0	11.3	1950	48
				18.4	2.6	.0	7.3	4.4	.0	11.3		
				6.4	1.9	.0	2.6	3.3	0	11.3		
Nic-0214	06031998	900	47.6	38.2	56.7	35.6	17.3	110.7	40.1	13.2	1951	47
				6.9	2.0	29.2	3.1	3.9	31.9	13.2		
				6.1	2.3	30.0	2.8	4.4	32.9	13.2		
Nic-0215	06021998	1200	375	.0	.0	0	.0	.0	.0	11.0	<1940	>58
				.0	.0	0	.0	.0	.0	11.0		
				880.4	341.8	66.6	355.3	600.2	64.1	11.0		

Appendix 2a. Chlorofluorocarbon data and apparent ground-water-recharge dates for 24 wells sampled in the low-sulfur coal region downgradient from reclaimed surface coal mines.—Continued

[ft, feet; pg/kg, picograms per kilogram; pptv, parts per trillion per volume; <, less than date or age; >, greater than date or age. CFC sample collected for well Boc-0257, however, no data tabulated here due to sample degradation and inconclusive results.]

Local county identifier	Date	Time	Well depth (ft below land surface)	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
Nic-0211	06101998	900	80	25.7	27.2	2.6	10.9	45.1	2.4	9.8	1961 with younger water	37
				26.0	22.5	2.7	9.8	37.3	2.4	9.8		
				19.4	21.8	.0	7.3	36.2	.0	9.8		
Nic-0219	06041998	900	57	4.4	3.2	.0	1.5	4.8	.0	7.5	1950	48
				4.1	2.4	.0	1.4	3.6	.0	7.5		
				9.5	4.2	.0	3.2	6.3	.0	7.5		
Nic-0217	06121998	1000	100	37.0	23.3	.0	13.4	37.1	.0	9.0	1961	37
				36.7	24.8	2.3	13.3	39.5	2.0	9.0		
				41.4	22.1	.0	15.0	35.2	.0	9.0		
Nic-0218	06171998	1100	90	292.4	226.0	38.3	130.6	434.6	41.2	12.8	1986	12
				273.5	226.2	35.4	122.1	435.0	38.1	12.8		
				277.8	230.9	33.7	124.1	444.0	36.2	12.8		

Appendix 2b. Chlorofluorocarbon data and apparent ground-water-recharge dates for 30 wells sampled in the high-sulfur coal region downgradient from reclaimed surface coal mines.

[pg/kg, picograms per kilogram; pptv, parts per trillion per volume]

Local county identifier	Date	Time	Well depth, in feet below land surface)	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
SO851	09171997	800	44	3.0	4.3	0.0	1.1	7.0	0.0	9.6	1951	46
				.1	3.3	.0	.0	5.3	.0	9.6		
				1.4	4.6	.0	.5	7.5	.0	9.6		
BT407	10161997	800	65	527.4	256.5	128.7	171.7	369.8	98.3	8.0	1984	13
				534.8	260.0	119.2	174.1	374.8	91.1	8.0		
				555.9	254.3	111.3	180.9	366.6	85.1	8.0		
JE422	08261997	800	60	40.1	32.9	.0	13.1	18.6	.0	8.0	1957	40
				41.5	15.1	.0	13.5	21.8	.0	8.0		
				37.2	14.7	.0	12.1	21.2	.0	8.0		
SO853	09161997	800	115	78.9	88.9	67.6	26.1	130.5	52.3	7.5	1966 with younger water	31
				69.7	79.1	18.1	23.0	116.1	14.0	7.5		
				71.9	83.9	40.7	23.8	123.2	31.5	7.5		
BA012	10081997	800	90	3.6	8,402.2	.0	1.3	13,256.9	.0	9.4	1953	44
				4.4	8,358.4	.0	1.6	13,187.9	.0	9.4		
				4.9	3,257.8	.0	1.8	5,140.1	.0	9.4		
PR773	09241997	800	41	440.3	571.3	47.1	155.1	886.3	39.2	9.0	1982	15
				442.2	533.4	39.7	155.8	796.5	33.0	9.0		
				432.2	482.5	45.6	152.3	748.6	37.9	9.0		
CA457	09151997	800	100	592.6	290.4	103.3	197.9	429.8	81.0	7.9	1988	9
				549.3	262.1	70.0	183.4	387.9	54.8	7.9		
				603.5	311.0	75.7	201.5	460.4	59.3	7.9		

Appendix 2b. Chlorofluorocarbon data and apparent ground-water-recharge dates for 30 wells sampled in the high-sulfur coal region downgradient from reclaimed surface coal mines.—Continued

[pg/kg, picograms per kilogram; pptv, parts per trillion per volume]

Local county identifier	Date	Time	Well depth, in feet below land surface)	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
WE313	08201997	800	135	0.0	0.0	0.0	0.0	0.0	0.0	8.5	1948	49
				1.9	2.6	.0	.6	3.8	.0	8.5		
SO852	08141997	800	70	.0	.0	.0	.0	.0	.0	9.8	1948	49
				10.2	80.3	5.0	3.8	130.6	4.4	9.8		
				.0	.0	.0	.0	.0	.0	9.8		
MN612	09231997	800	40.7	100.1	443.3	.0	40.6	778.4	.0	12.0	1970	27
				101.1	362.9	.0	41.0	637.1	.0	12.0		
				110.7	373.0	.0	44.9	654.9	.0	12.0		
JE424	09101997	800	50	3.6	1,529.5	.0	1.2	2,336.4	.0	8.6	1952	45
				.6	1,506.4	.0	.2	2,303.1	.0	8.6		
				1.1	1,502.0	.0	.4	2,296.4	.0	8.6		
JN917	09031997	800	82	.0	.0	9.5	.0	.0	7.4	8.3	1945	52
				.0	.0	10.3	.0	.0	8.0	8.3		
				.0	1.4	.0	.0	2.0	.0	8.3		
BT406	08191997	800	68	384.5	528.2	5.8	112.3	690.6	4.0	6.3	1973	24
				346.7	432.1	7.5	101.3	564.9	5.1	6.3		
				338.1	387.4	6.9	98.8	506.4	4.7	6.3		
JN918	09021997	800	65	4.6	3.7	.0	1.5	5.4	.0	8.2	1947	50
				.0	.0	.0	.0	.0	.0	8.2		
				.2	.0	.0	.1	.0	.0	8.2		

Appendix 2b. Chlorofluorocarbon data and apparent ground-water-recharge dates for 30 wells sampled in the high-sulfur coal region downgradient from reclaimed surface coal mines.—Continued

[pg/kg, picograms per kilogram; pptv, parts per trillion per volume]

Local county identifier	Date	Time	Well depth, in feet below land surface)	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
AR107	08271997	800	154	5,810.1	3,101.5	17.6	1,953.9	4,608.8	13.9	8.4	1977	20
				4,915.1	6,354.5	18.8	1,953.0	9,442.9	14.9	8.4		
				5,891.1	2,902.3	16.9	1,981.2	4,312.9	13.4	8.4		
WS973	09181997	800	98	109.0	67.2	.0	35.5	97.0	.0	8.1	1967	30
				84.0	63.8	.0	27.4	92.1	.0	8.1		
				85.3	64.1	8.4	27.8	92.5	6.4	8.1		
UP972	10201997	800	68	131.5	63.2	.0	48.9	102.7	.0	10.2	1971 with older water	26
				137.4	72.4	.0	51.1	117.7	.0	10.2		
				150.5	66.1	.0	56.0	107.4	.0	10.2		
IN861	10011997	800	85	98.0	92.0	28.6	32.4	134.5	22.2	8.0	1969 with younger water	28
				92.1	89.8	25.6	30.4	131.4	19.9	8.0		
				154.3	90.5	19.2	51.0	132.4	14.9	8.0		
BA011	10211997	800	52	52.4	40.9	.0	19.5	66.5	.0	10.2	1966	31
				75.4	55.1	.0	28.1	89.7	.0	10.2		
				81.2	56.5	34.3	230.2	92.0	30.3	10.2		
WE311	09251997	800	220	510.6	284.8	75.4	166.7	411.9	57.8	8.0	1987	10
				507.9	275.6	68.1	165.9	398.5	52.2	8.0		
				539.6	265.9	87.2	169.6	384.5	66.9	8.0		
FA508	08211997	800	135	415.4	562.2	52.9	138.8	831.3	41.5	8.2	1977 with younger water	20
				426.3	571.3	50.7	139.2	844.8	39.8	8.2		
				404.1	551.0	43.5	135.1	824.8	34.2	8.2		

Appendix 2b. Chlorofluorocarbon data and apparent ground-water-recharge dates for 30 wells sampled in the high-sulfur coal region downgradient from reclaimed surface coal mines.—Continued

[pg/kg, picograms per kilogram; ppt, parts per trillion per volume]

Local county identifier	Date	Time	Well depth, in feet below land surface	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
HR332	09301997	800	96	47.5	57.8	5.6	19.6	102.8	5.5	12.4	1963 with older water	34
				6.0	36.5	.0	2.5	64.9	.0	12.4		
				26.0	32.0	.0	10.7	55.2	.0	12.4		
PR774	09291997	800	110	28.7	36.7	7.2	10.9	60.9	6.5	10.7	1967	30
				30.7	39.4	3.2	11.7	65.4	2.9	10.7		
				36.6	54.5	8.3	13.9	90.3	7.5	10.7		
AR106	08281997	800	55	2.7	.0	.0	.9	.0	.0	9.0	1953	44
				4.9	6.0	.0	1.7	9.0	.0	9.0		
				4.4	1.0	.0	1.5	1.5	.0	9.0		
WE314	10021997	800	140	4.6	29.7	.0	1.6	44.9	.0	9.0	1962	35
				4.0	27.1	.0	1.4	40.9	.0	9.0		
				4.4	27.5	.0	1.5	41.6	.0	9.0		
LE411	10091997	800	85	33.9	22.0	6.8	13.2	37.2	6.4	11.3	1961 with younger water	36
				34.6	41.7	8.8	13.5	70.6	8.2	11.3		
				17.6	10.3	11.7	6.9	17.4	10.9	11.3		
WE312	08071997	800	150	154.2	116.9	6.3	51.6	172.6	4.9	8.5	1976	21
				154.8	113.7	25.9	51.8	167.9	20.4	8.5		
				150.9	122.2	24.2	50.5	180.5	19.0	8.5		
VE058	09081997	800	63	43.6	549.4	30.4	14.9	827.7	24.5	8.7	1962 with younger water	35
				37.8	519.7	25.1	12.9	782.9	20.2	8.7		
				41.5	570.1	28.1	14.2	858.8	22.6	8.7		

Appendix 2. Overall and concentration-weighted pesticide-mixture scores for 28 wells sampled in the high-sulfate regional comparison for human health and aquatic-life—Continued

[Only the samples with a higher score for either human health or aquatic life are shown. Blank cells indicate no data]

Local county identifier	Date	Time	Total pesticides, in the human health and aquatic-life mixtures	Overall mixture score, in water year			Overall mixture score, in calendar year			Percentage human-health mixture	Other human-health mixture score, approved OR aquatic-life mixture	Selected age of water based on tritium
				CY-1	CY-2	CY-3	CY-1	CY-2	CY-3			
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
				[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		
				[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
				[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		
				[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		

Appendix 2c. Chlorofluorocarbon data and apparent ground-water-recharge dates for 5 wells sampled in unmined areas of the low-sulfur coal region.

[pg/kg, picograms per kilogram; pptv, parts per trillion per volume]

Local county identifier	Date	Time	Well depth, in feet below land surface	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluoro-carbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
Ral-0220	05131997	900	226	8,417.8	269.5	59.2	3,063.7	431.6	51.5	9.1	1986	11
				8,426.5	260.1	51.0	3,066.9	416.6	44.4	9.1		
				8,252.8	235.5	47.3	3,003.7	377.2	41.1	9.1		
Kan-0927	05281997	1100	65	100.5	89.8	17.3	34.7	122.7	14.4	9.4	1970 with younger water	27
				114.1	91.5	20.8	39.5	139.0	17.2	9.4		
				151.6	118.0	28.1	52.4	179.2	23.3	9.4		
Cla-0141	05082997	1100	50	65.7	88.3	24.8	25.4	148.6	23.2	11.2	1972	25
				51.5	69.4	8.4	19.9	116.7	7.9	11.2		
FA509	07221998	800	200	238.2	487.1	34.3	58.2	788.0	30.2	10.2	1973	25
				244.5	445.9	32.6	90.5	721.4	28.6	10.2		
				240.2	485.0	32.7	63.9	784.6	28.7	10.2		
LE412	08271998	800	112	27.4	208.5	40.9	10.8	356.1	38.4	11.3	Unknown	Unknown
				21.4	165.7	57.1	4.5	282.9	53.7	11.3		
				47.6	142.0	57.8	18.7	242.4	54.3	11.3		

Appendix 2d. Chlorofluorocarbon data and apparent ground-water-recharge dates for 9 wells sampled in unmined areas in the High-sulfur coal region.

[pg/kg, picograms per kilogram; pptv, parts per trillion per volume]

Local county identifier	Date	Time	Well depth, in feet below land surface	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
Nic-0208	06161997	1100	208	7.9	14.9	0.0	2.2	19.1	0.0	5.3	1955	42
				9.1	11.5	.0	2.6	14.8	.0	5.3		
				45.7	36.2	.0	13.0	46.6	.0	5.3		
SO124	07111996	800	75	2.7	48,255.0	.0	1.0	79,112.1	0	11.2	1947	50
				.6	43,737.4	.0	.2	71,705.8	0	11.2		
				.7	45,188.7	.0	.3	74,085.0	.0	11.2		
SO126	07031996	800	114	295.4	253.4	59.4	115.6	429.9	56.4	11.6	1987	10
				297.0	252.2	60.4	116.2	428.0	57.3	11.6		
				309.3	266.4	69.6	121.0	452.1	66.1	11.6		
DN915	08011996	800	95	339.8	201.6	47.5	136.6	351.0	46.4	11.8	1983	14
CA456	08201996	800	72	1.0	.0	.0	.3	.0	.0	10.0	1947	50
				.7	2.3	.0	.2	3.6	.0	10.0		
				1.2	1.8	4.3	.4	2.8	3.7	10.0		
Bro-0269	06301997	1100	90	1.7	4.9	.0	.6	7.3	.0	8.7	1950	47
				2.5	3.8	.0	.8	5.7	.0	8.7		
				2.2	2.0	.0	.7	3.0	.0	8.7		
CR727	07161998	800	178	9.8	10.0	.0	3.4	15.1	.0	8.7	1954	44
				7.6	12.1	.0	2.6	18.3	.0	8.7		
				9.4	6.9	.0	3.2	10.4	.0	8.7		

Appendix 2d. Chlorofluorocarbon data and apparent ground-water-recharge dates for 9 wells sampled in unmined areas in the high-sulfur coal region.—Continued

[pg/kg, picograms per kilogram; pptv, parts per trillion per volume]

Local county identifier	Date	Time	Well depth, in feet below land surface	Chlorofluorocarbon concentration in water (pg/kg)			Chlorofluorocarbon partial pressure (pptv)			Recharge temperature	Chlorofluorocarbon apparent CFC recharge date	Estimated age of ground water (years)
				CFC-11	CFC-12	CFC-113	CFC-11	CFC-12	CFC-113			
AR108	07291998	800	100	11.4	4.9	0.0	3.9	7.3	0.0	8.7	1952	46
				13.7	10.6	.0	4.6	15.9	.0	8.7		
				10.0	6.2	.0	3.4	9.3	0	8.7		
WE315	09081998	800	30	478.6	292.1	43.0	166.6	447.3	35.3	9.1	1986	12
				504.1	265.1	47.2	175.4	405.9	38.8	9.1		
				507.1	307.1	44.3	176.5	470.2	36.4	9.1		

Appendix 3. Site identification numbers, local well names, coal-sulfur-content region, land-use classification, topographic location, distance from mine, and well depth for wells sampled.

[NA, not applicable]

Site identification number	Local well name	Coal sulfur content region H = High L = Low	Land-use classification	Topographic location	Distance from mine, in feet[1]	Well depth, in feet below land surface
390201080311301	LE411	H	downgradient from mine	Valley	197	85
390227080145801	UP972	H	downgradient from mine	Valley	492	68
390353079555601	BA012	H	downgradient from mine	Hillside	26	90
391103079582901	BA011	H	downgradient from mine	Hillside	394	52
392458080165301	HR332	H	downgradient from mine	Valley	98	96
392917079460501	PR773	H	downgradient from mine	Hillside	20	41
392924079374601	PR774	H	downgradient from mine	Hillside	0	110
393533080035501	MN612	H	downgradient from mine	Hillside	49	46
395040078595401	SO851	H	downgradient from mine	Hillside	1,509	44
395520079080301	SO852	H	downgradient from mine	Hillside	26	70
400434079235201	FA508	H	downgradient from mine	Hillside	0	135
400455078532101	SO853	H	downgradient from mine	Hillside	394	115
400846079485701	WE312	H	downgradient from mine	Hillside	951	150
401053079590001	WS973	H	downgradient from mine	Hillside	394	98
401225079361501	WE313	H	downgradient from mine	Hillside	98	135
402319079283001	WE314	H	downgradient from mine	Hilltop	0	140
402415078544301	CA457	H	downgradient from mine	Hillside	984	100
403253079410701	WE311	H	downgradient from mine	Hillside	0	220
403650079220201	IN917	H	downgradient from mine	Valley	328	82
403756079234001	AR106	H	downgradient from mine	Valley	492	55
404050079494001	BT406	H	downgradient from mine	Hillside	98	32
404230078540901	IN861	H	downgradient from mine	Valley	492	135
404712078575501	IN918	H	downgradient from mine	Hillside	197	65
405449079193901	AR107	H	downgradient from mine	Hillside	98	154
405642079455601	BT407	H	downgradient from mine	Valley	98	65
410156079183801	CR726	H	downgradient from mine	Hillside	49	70
410205079061101	JE422	H	downgradient from mine	Hillside	295	60
410609078505701	JE423	H	downgradient from mine	Hillside	295	98
411026078484301	JE424	H	downgradient from mine	Valley	295	50
411336079574901	VE058	H	downgradient from mine	Hillside	197	63
385131080272501	LE412	H	in unmined area	Hillside	NA	112
395200079160101	SO124	H	in unmined area	Hilltop	NA	75
395201079383601	FA509	L	in unmined area	Hilltop	NA	200
400953079382101	WE309	H	in unmined area	Hillside	NA	55
401102079273801	WE310	H	in unmined area	Hillside	NA	250
401337079005201	SO126	H	in unmined area	Valley	NA	114
402629079092301	IN915	H	in unmined area	Hillside	NA	95
402905079400001	WE315	H	in unmined area	Hillside	NA	30
402934079211401	IN914	H	in unmined area	Hillside	NA	160
403237078463101	CA456	H	in unmined area	Hillside	NA	72

Appendix 3. Site identification numbers, local well names, coal-sulfur-content region, land-use classification, topographic location, distance from mine, and well depth for wells sampled.—Continued

[NA, not applicable]

Site identification number	Local well name	Coal sulfur content region H = High L = Low	Land-use classification	Topographic location	Distance from mine, in feet[1]	Well depth, in feet below land surface
404108079273001	AR105	H	in unmined area	Hillside	NA	110
404510079104101	IN913	H	in unmined area	Hillside	NA	123
404924079155201	AR108	H	in unmined area	Hillside	NA	100
410557078465901	CF320	H	in unmined area	Hillside	NA	110
411335079375301	CR727	H	in unmined area	Hilltop	NA	178
375115080474901	Sum-0105	L	downgradient from mine	Hillside	1,148	115
375246081133401	Ral-0221	L	downgradient from mine	Hillside	984	49.5
375420081091301	Fay-0274	L	downgradient from mine	Hillside	164	47.5
375739081171801	Fay-0275	L	downgradient from mine	Valley	820	21.3
375829081180801	Fay-0276	L	downgradient from mine	Valley	656	102
375914081121501	Fay-0278	L	downgradient from mine	Hillside	328	42.7
375915081132501	Fay-0279	L	downgradient from mine	Valley	1,476	30
380106080432801	Grb-0284	L	downgradient from mine	Hillside	1,804	112
380131080591301	Fay-0272	L	downgradient from mine	Hillside	295	225
380153081341101	Boo-0260	L	downgradient from mine	Valley	820	73
380444081351401	Boo-0258	L	downgradient from mine	Valley	1,640	64
380607081145801	Fay-0270	L	downgradient from mine	Valley	3,117	27
380708081370201	Boo-0259	L	downgradient from mine	Hillside	2,133	105
380818081502301	Boo-0255	L	downgradient from mine	Valley	1,476	200
380939081504801	Boo-0256	L	downgradient from mine	Hillside	328	130
381048081504801	Boo-0257	L	downgradient from mine	Valley	1,148	120
381125081304701	Kan-0935	L	downgradient from mine	Hillside	1,312	100
381513080584401	Nic-0212	L	downgradient from mine	Valley	492	25.5
381539080560201	Nic-0209	L	downgradient from mine	Valley	328	50
381656080543301	Nic-0213	L	downgradient from mine	Valley	984	48.4
381847081254201	Kan-0938	L	downgradient from mine	Valley	984	105
381931080425401	Nic-0214	L	downgradient from mine	Valley	656	47.6
382011080424301	Nic-0215	L	downgradient from mine	Valley	328	375
382113080442701	Nic-0211	L	downgradient from mine	Hillside	1,148	80
382123080381701	Nic-0219	L	downgradient from mine	Valley	131	57
382307080381201	Web-0238	L	downgradient from mine	Valley	230	26.5
382340080481301	Nic-0217	L	downgradient from mine	Valley	492	100
382434080401401	Nic-0218	L	downgradient from mine	Hillside	262	90
374645081030701	Ral-0220	L	in unmined area	Hilltop	NA	226
381639080380101	Nic-0206	L	in unmined area	Valley	NA	115
381652080473501	Nic-0208	H	in unmined area	Hillside	NA	208
381852081404401	Kan-0927	L	in unmined area	Hillside	NA	65
382131081091501	Cla-0141	L	in unmined area	Valley	NA	50
382636081432801	Kan-0928	H	in unmined area	Valley	NA	73
382845081300301	Kan-0932	H	in unmined area	Hillside	NA	75

Appendix 1. Site identification numbers, local well names, coastal region, land-use classification, topographic position, distance from mine, and total depth to water surface—Continued

[N/A, not applicable]

Site identification number	Local well name	Coastal region (H = High, L = Low)	Land-use classification	Topographic position	Distance from mine (km)[1]	Total depth to water below land surface
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	N/A	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	Valley	N/A	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	Valley	N/A	[illegible]

[1]Distance from mine is not applicable for non-mine locations and was assigned a value of zero for the analysis.

EXHIBIT 9

Ecological Studies

Volumes published since 2002 are listed at the end of this book.

R. Bobbink B. Beltman J.T.A. Verhoeven D.F. Whigham (Eds.)

Wetlands: Functioning, Biodiversity Conservation, and Restoration

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Ecological Studies, Vol. 191

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Cover illustration: Large photo: Overview of an undisturbed tidal freshwater wetland system at Jug Bay Sanctuary on the Patuxent River, Maryland, USA (Photo R. Bobbink). Small photos: *Top* Dense beds of the invasive species *Crassostrea gigas*, the Pacific oyster, near Yerseke, The Netherlands, after intentional introduction of this species for cultivation (Photo D.W. Thieltges). *Middle* Restoration of a degraded rich fen by removal of the vegetation and the acidified top layer of the peat, The Netherlands (Photo B. Beltman). *Bottom* Extensive aerenchyma in the stem of pond weed (*Potamogeton pectinatus*) (Photo by Summers and Jackson 1994).

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Preface

The two volumes on "Wetlands as a Natural Resource" in the book series Ecological Studies (Volumes 190, 191) are based on the highlights of the 7th INTECOL International Wetland Conference in Utrecht, 25–30 July 2004. This conference brought together about 900 participants from 81 countries, who discussed a very broad range of science-, policy- and management-oriented issues related to wetland ecology and hydrology, wetland conservation and creation, the impact of global change and wetlands as a resource in terms of food, flood protection and water quality enhancement. The participants were from different sectors of society, i.e., science and technology (scientists 45 %; PhD students 20 %), natural resource management (20 %) and policy (15 %). There were 38 symposia with invited speakers centered around the nine conference themes. We have given the organizers of these symposia the opportunity to produce one chapter for these books with the integrated content of their symposium. This has resulted in 25 chapters, of which 13 are included in Volume 190 under the heading "Wetlands and Natural Resource Management" and 12 in Volume 191 under the heading "Wetlands: Functioning, Biodiversity Conservation, and Restoration".

With these books, we had the aim to summarise the most important recent scientific results in wetland science, their applications in wetland and water resource management and their implications for the development of global, national and regional policies in the perspective of the ever-progressing deterioration of natural wetlands and the major impacts that future climate change will have. We hope that the integrated content of the chapters on such a wide scope of different fields in wetland science will serve as a valuable source of information, both for professionals in environmental science and natural resource management and for students and young professionals seeking to familiarize themselves with these fields. We also hope that the interaction between scientists from different disciplines, resource managers and policy makers will be stimulated by the content of these publications.

We as editors have worked according to a strict time schedule and we want to thank the authors for their timeliness in producing inspiring manuscripts and the scientists who have contributed to the peer reviews of the chapters for their active and prompt participation, which has enabled us to complete our task more or less according to this schedule. We acknowledge the series editor of the Ecological Studies book series, Prof. Dr. Ulrich Sommer, for his invitation to produce these volumes as one of the outcomes of the INTECOL Conference. We also thank Dr. Andrea Schlitzberger of Springer for her advice and help. We would like to take the opportunity to thank all key people who made the conference into such a success. In particular we want to thank Prof. Dr. Eugene Turner and the other members of the INTECOL Wetlands Working group, as well as the

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11 Fen Management and Research Perspectives: An Overview

Beth Middleton, Ab Grootjans, Kai Jensen, Harry Olde Venterink, and Katalin Margóczi

11.1 Introduction

A fen has vegetation that is actively forming peat and is fed by ground- or surface water (Joosten and Clarke 2002). In Europe a "fen meadow" is a ground- or surface water-fed mown grassland that does not form peat, since it was formed after modest drainage of a fen or is developed on a predominantly moist soil (Grootjans and Van Diggelen 1995). Therefore, fens and fen meadows are considered to be different ecosystems by most European authors. Others do not make a distinction between fens and fen meadows because the species composition of both ecosystems may overlap considerably (Wheeler et al. 1995). In North America, fens dominated by tussock-forming sedges are referred to as "sedge meadows", which are often grazed. Since there is not yet scientific agreement on whether sedge meadows are fens or fen meadows, we will refer to them as fens in this chapter.

Fens are the most diverse temperate community types (Bedford and Godwin 2003). The biodiversity of fens is threatened by agricultural and urban development, fragmentation, hydrologic changes, shifts in grazing and mowing, structural changes in vegetation, and reproductive problems related to isolated populations. Species are declining in numbers in fens in Europe and North America. Estimates of species lost, for example, from the United Kingdom, are that 95–98 % of species-rich fens present before 1940 have been lost (Garcia 1992). Similar losses are reported for France (Muller et al. 1998) and the Netherlands (Jansen et al. 2000).

The fragmentation of fens occurs via agricultural and urban development; and as these encroach, the wetlands become smaller and more isolated because of hydrological modification (Figs. 11.1, 11.2). In particular, any alteration in surface water flow from streams and rivers creates a barrier to seed dispersal (Galatowitsch and van der Valk 1996; Middleton 1999). Species loss

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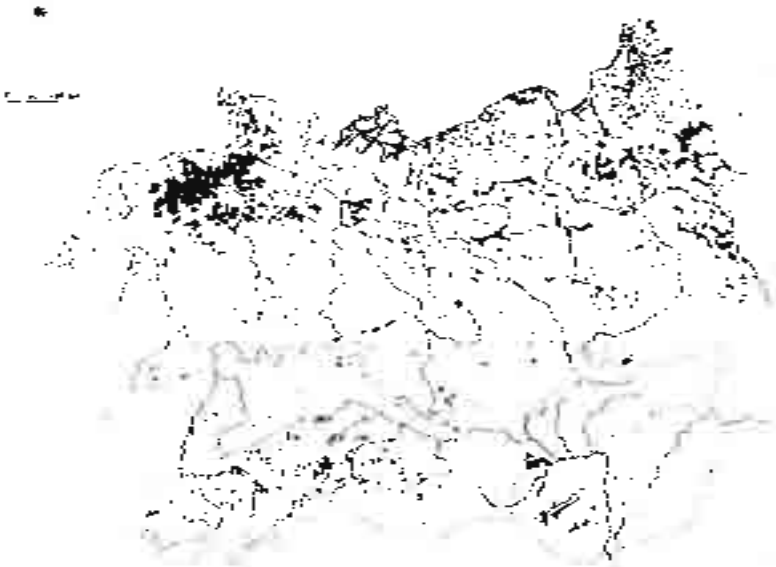

Fig. 11.1 Mire distribution in Central Europe under natural conditions (Succow 1988; reproduced by permission of Fischer Verlag, Jena). Only peatlands exceeding 300 ha have been included. Black 3D bogs, gray 3D fens

occurs because of the lack of habitat (Bedford and Godwin 2003), but also because of reproductive problems associated with small population sizes (Ellstrand and Elam 1993; Fischer and Stöcklin 1997). While these problems might be alleviated by the reconnection of dispersal corridors (Taylor et al. 1993), higher amounts of dispersal between these disconnected wetlands could contribute to more opportunities for an invasion of exotic species in North America, where invasive species in wetlands are a major problem (Galatowitsch et al. 1999).

Water quality and sedimentation patterns can be altered by agricultural and urban development; and these changes contribute to shifts in species composition in fens. Sedimentation is on the increase in fens adjacent to cities in the Midwestern United States; and this encourages invasive species (*Phalaris arundinacea*, *Typha* x *glauca*) at the expense of *Carex stricta* (Werner and Zedler 2002). Urban water runoff increases the amounts of N and P in the surface water, and subsequently, the biomass of *T.* x *glauca* increases. *C. stricta* does not increase in biomass, so that fens with urban runoff become dominated by *T.* x *glauca* (Woo and Zedler 2002). The nutrient dynamics in fens changed with land use changes. Atmospheric nitrogen

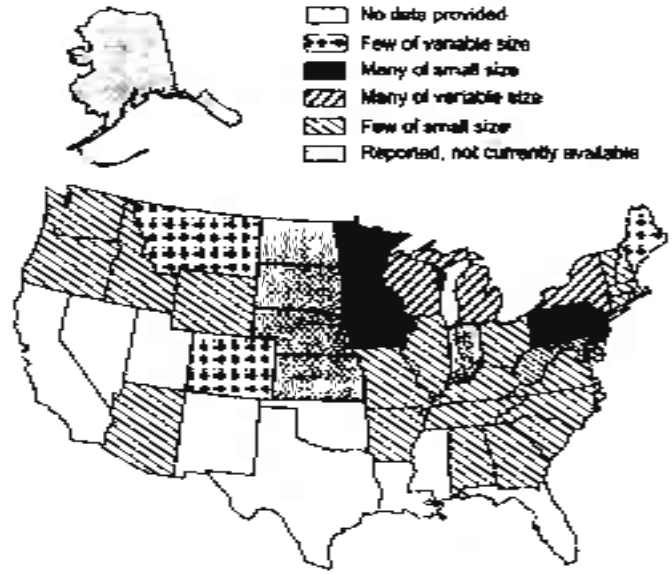

Fig. 11.2 Distribution of North American fens (Bedford and Godwin 2003; reproduced by permission of Society of Wetland Scientists)

deposition, in particular, increased 3- to 10-fold during the past decades in these wetlands (Olde Venterink et al. 2001, 2002). Eutrophication causes increases in the plant biomass of certain species and contributes to the encroachment of large herbaceous and woody species (Bobbink and Roelofs 1995).

11.2 Hydrological Systems of Fens

Fens with similar vegetation types exist in very different landscapes and soil conditions. Small sedge vegetation with many Red List species (*Caricion davallianae*, i.e., species of alkaline fens), for instance, occur in coastal dune slacks on sandy soils (Ranwell 1959), in brook valleys of the European lowlands on peat soils (Succow and Joosten 2001), but also in calcareous spring mires in alpine regions of Europe (Ellenberg 1986).

Hydrological systems can create similar habitats within a wide range of landscapes. Groundwater fed fens are "flow through" systems, in which nutri-

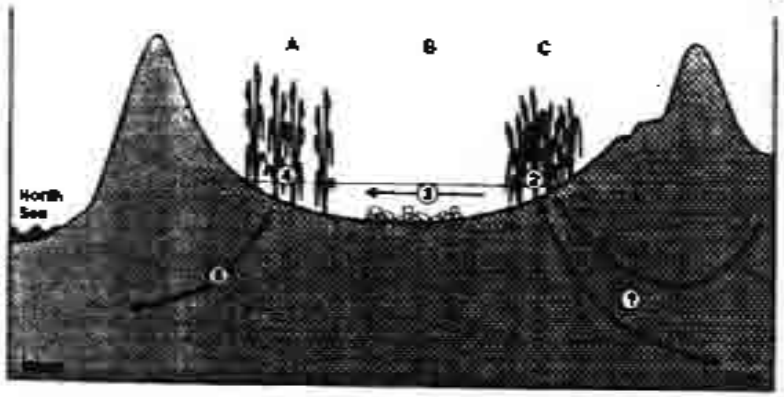

Fig. 11.3 Schematic presentation of a through flow dune slack on the Dutch Wadden Sea island. 1 Incoming calcium and iron-rich groundwater, 2 infiltration of groundwater in a vegetation zone with sedges, 3 surface flow and precipitation of iron and $CaCO_3$ in a zone where nutrient-poor pioneer vegetation persists, 4 infiltration of iron- and calcium-poor surface water in a *Phragmites australis* stand, 5 sulfate reduction during infiltration (after Adema 2002; reproduced by permission of Opulus Press)

ents become less available for plant growth due to geochemical changes along the flow paths of the groundwater (Fig. 11.3).

11.2.1 Large and Small Hydrological Systems

Much research has been done on the hydrological conditions of fen systems (Romer 1994; Wassen et al. 1996; Van Diggelen 1998; Jansen et al. 2000; Schot et al. 2004). Some authors present very detailed hydrological research in species-rich fens on mineral soils, a case in which most of the groundwater comes from a very small catchment area (Jansen et al. 2000), while others worked in very large mire systems, like the Biebrza in Poland (Wassen et al. 1996) and the Ob River in Russia (Loos and Schipper 2003; Fig. 11.4). Sometimes the surface area of a fen is rather small, but the catchment area supplying the fen with groundwater is very large. This is often the case with fens and spring mires that are supplied with artesian water, originating from large aquifers in which the water flow has been blocked due to shifting of geological strata (Van Andel and Aronson 2005). Fens are very vulnerable to erosion by groundwater flow. The digging of drainage ditches may result in the downcutting of the ditch channel, which effectively lowers the local water table within the fen. The peat may begin to decompose and erode into the ditch. After the ditch is established, the degradation process may be very difficult to

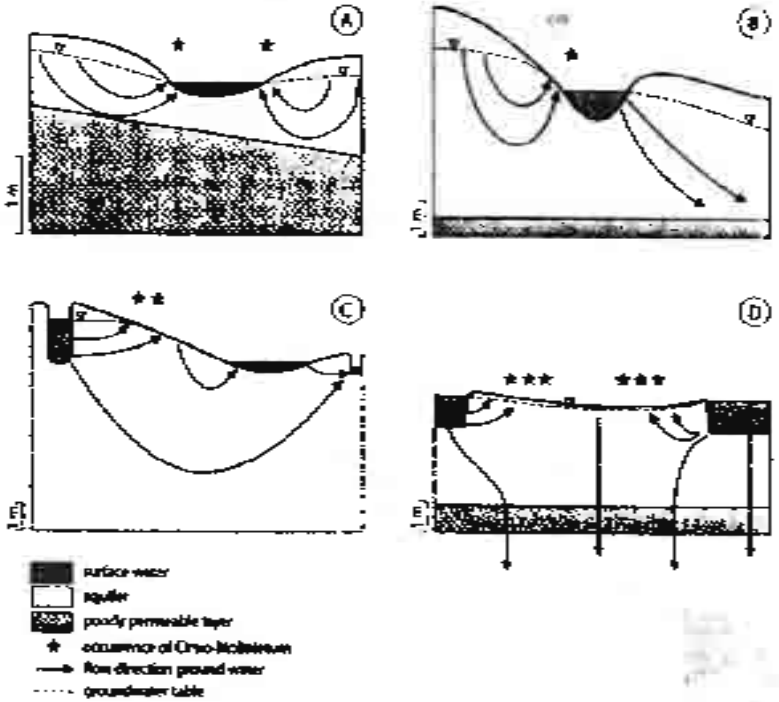

Fig. 11.4 The fen meadow community *Cirsio-Molinietum* (indicated by stars) occurs in the Netherlands under various hydrological conditions. A Heathland pool situated above shallow loam layer with groundwater supply from local sand ridges, B local (small) hydrological system with through flow of groundwater, C artificial hydrological system with lateral discharge of calcareous surface water from large canals, D surface water system with discharge of calcareous surface water from ditches (modified after Jansen et al. 2000; reproduced by permission of Opulus Press)

contain, and may result in the total destruction of the mire (Wolejko et al. 1994). Due to rapid peat mineralization, these systems become very eutrophic and tall herbs (such as *Urtica dioica*), shrubs, and trees move in quickly (Wolejko et al. 1994).

11.2.2 Natural Fens Can Be Very Stable

Fens are mires that receive water from the mineral surroundings and hence respond to changes in regional hydrology. Large fens, however, also raise their

water level and that of their surroundings autogenously, leading to the expansion of fens on progressively higher positions in the landscape. Such rising water levels may also result in the gradual paludification of hitherto dry upland depressions and may even reverse water flow from groundwater recharge to groundwater discharge conditions (Succow and Joosten 2001).

Due to their thick and rather flexible peat layer, such fens became increasingly more resilient during the course of their long-term development and remained in a more or less stable situation for several centuries, despite considerable climatological and hydrologic changes. They only changed character when certain hydrological thresholds were crossed, the exact magnitude of these thresholds being dependent on the stage of mire development. Schult (2002) carried out detailed palynological research in the Trebel area and found that between 8000 and 7000 years BP, a terrestrialization mire developed in a calcareous lake, with species such as *Menyanthes trifoliata*, *Cladium mariscus*, and *Carex rostrata*. At this stage, small amounts of $CaCO_3$ (travertine) were deposited. Between 7000 BP and 5000 BP, a groundwater-fed fen developed on top of this young mire, with predominantly moss species, such as *Drepanocladus* spp, *Homalothecium nitens*, and *Meesia triquetra*. *Menyan-*

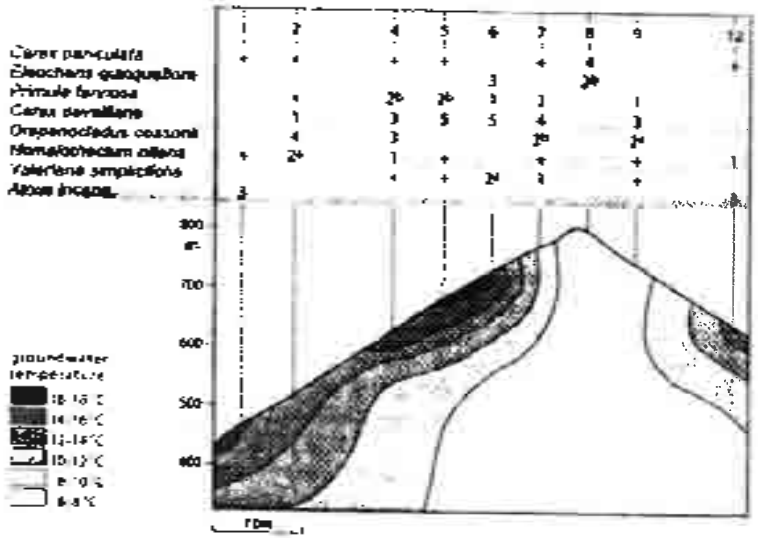

Fig. 11.5 Temperature profile and vegetation composition across a small Slovak spring mire near the High Tatra Mountains. The species coverage was estimated using a Braun-Blanquet cover/abundance scale (Grootjans, unpublished data)

thes trifoliata remained present throughout this period and small sedges such as *C. diandra* and *C. limosa* also occurred. Chalk deposition occurred until ca. 3000 BP and then stopped. This developmental sequence was remarkably stable, and similar results have been found in Western Germany (Schwaar 1980) and Poland (Wassen and Joosten 1996).

Nowadays such vegetation types are very rare and chalk deposition is almost no longer found in the North German lowlands. However, remnants of these vegetation types can still be found in the mountain areas of Bavaria (Germany) and Slovakia. Here we find relatively small calcareous spring mires, where travertine is still being formed in pools with *Drepanocladus* spp, *Menyanthes trifoliata*, and *Chara* spp. This travertine formation is a rather recent phenomenon in some wetlands and mainly occurs in oligotrophic (calcareous) pools where supersaturated groundwater discharges (Grootjans et al. 2005). During summer the surface water of these pools is warmed (Fig. 11.5). Outgassing occurs and $CaCO_3$ precipitates. Especially moss species and some water plants, such as *Chara* spp, can accelerate this travertine formation by using the high concentration of CO_2 in the pools as a carbon source. $CaCO_3$ precipitates on the leaves and is later added to the soil (Van Breemen and Buurman 2002). The added travertine prevents acidification and associated eutrophication in the soil and sustains a very high biodiversity in such fens. However, such small groundwater-fed systems are vulnerable to hydrological changes in the landscape and also to climatic change. These small groundwater-fed fens can easily shift from peat-forming systems toward eroding stages within a short period of time (Wolejko et al. 1994).

11.2.5 Hydrochemical Processes Stabilizing the Biodiversity of Fens

Formerly, hydrological systems that provided natural fens with a large supply of base-rich groundwater were able to stabilize nutrient poor fen vegetation for many centuries, without any management by man. However, during the past few centuries, almost all fens have been slightly drained and changed into low-productive meadows and pastures that cannot be maintained without management (Kotowski 2002). When large hydrological changes in the surrounding landscape occur, the through-flow of anaerobic groundwater should be re-established to restore the high biodiversity of fen species. A relatively slow groundwater flow in fen systems is essential for maintaining a high biodiversity. Slow-flowing groundwater prevents erosion and stimulates the precipitation of iron and $CaCO_3$ in the root zone, thus stabilizing nutrient cycling at a low level (Lamers et al. 2002; Olde Venterink et al. 2001, 2002). The study of historical descriptions and well preserved remnants of fen systems may prove very helpful in understanding why some systems can easily maintain a high biodiversity and why other cannot perform this function.

11.3 Eutrophication in Fens

Eutrophication is considered one of the main factors contributing to the loss of biodiversity in fens in Western Europe, because increased nutrients are related to increased productivity and the competitive exclusion of shorter species. Species-rich communities of low productivity are eventually transformed into communities of high productivity dominated by a few tall sedges, grasses or shrubs (Grime 1979). This increased dominance of tall herbaceous and woody species occurred both in Europe and North America, not only due to eutrophication, but also because of changes in rural land usage related to grazing (Middleton 2002a) and drainage (Wheeler et al. 1995; Grootjans et al. 1996).

11.3.1 Change in Management

After 1945, agriculture shifted away from small farms and moved toward larger mechanized ones that no longer utilized fens for pastures (Middleton 2002a) or hay-mowing (Garcia 1992). After that, the short diverse vegetation shifted toward communities dominated by woody and tall herbaceous species. These changes in vegetation structure are directly attributable to the cessation of cattle grazing and mowing. Especially after moderate to heavy grazing, shrubs and trees grow and shade the rarer fen species (Middleton 2002a, b).

Strategies for dealing with these changes in community structure and related losses in biodiversity have taken different routes in Europe and North America. Managers in North America control the dominance of woody and tall species with fire, whereas European managers rely on mechanical mowing and hay removal (Schrauzer et al. 1996). Fire temporarily increases the species richness, flowering and seed set of fen species in North America (Middleton 2002b). Because eutrophication is a contributing factor in the growth of these large fen species, fire may also be useful because it removes nutrients such as nitrogen from the system. Similarly, hay removal also removes nutrients from these systems, which eventually may lead to phosphorus limitation, and is therefore of value in maintaining the biodiversity of fens (Verhoeven et al. 1996; Tallowin and Smith 2001).

11.3.2 Change in Nutrient Budgets

The increase in productivity in fens is the result of increased availabilities of potentially growth limiting nutrients such as nitrogen (N), phosphorus (P), and potassium (K; Verhoeven et al. 1996; Van Duren and Pegtel 2000). To

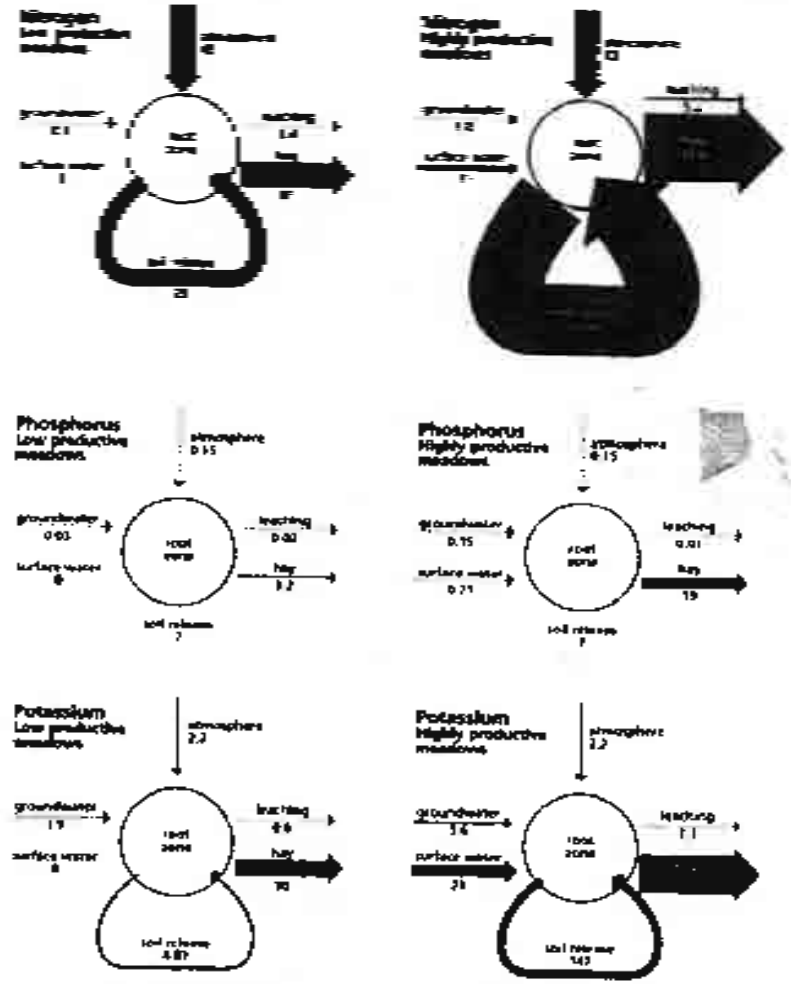

Fig. 11.6 Annual N, P, and K fluxes into, out of, and within the root zone of low-productive species-rich meadows and highly productive species-poor meadows. All fluxes are in kg N, P, or K per hectare per year (Olde Venterink et al. 2002; reproduced by permission of the Ecological Society of America)

understand the causes of nutrient enrichment and evaluate the effects of management, one should have insight into annual nutrient budgets, i.e., the rates of nutrient fluxes into and from external sources as well as net release rates of nutrients in the soil (Koerselman et al. 1990).

Olde Venterink et al. (2001, 2002) quantified N, P, and K budgets along productivity gradients in fens to evaluate the importance of atmospheric deposition, flooding, groundwater flow, leaching, and soil release rates. Differences in N availability along productivity gradients were caused by release rates in the soil (mineralization; Fig. 11.6). Besides N release in the soil, atmospheric N deposition made up large proportions of the annual N budgets. P and K availabilities along the productivity gradients were mainly influenced by soil processes, as indicated by soil extractable nutrient. The input and output fluxes of nutrients by groundwater flow were very low, while the input by flooding seemed only important for K (Fig. 11.6). However, it should be noted that the input of nutrients adsorbed to sediment was not included in the flooding assessment, which could be a very important input source of P in fens (Mitsch et al. 1979). The dominant role of soil processes in setting nutrient availabilities along the productivity gradients demonstrated that human alterations of site conditions, such as drainage, have been a major cause of nutrient enrichment for N, P, and K (Olde Venterink et al. 2002). Furthermore, considering that atmospheric N deposition has increased from 5–20 kg N ha^{-1} $year^{-1}$ (Erisman and Draaijers 1995) to an average 43 kg N ha^{-1} $year^{-1}$ in this part of Europe (Fig. 11.6), increased atmospheric N deposition also has been a major source of nutrient enrichment.

The long-term effects of annual hay-harvesting on nutrient limitation were also assessed by Olde Venterink et al. (2002). In fens with low productivity, N output by hay-harvesting accounted for N input from atmospheric deposition, whereas there was a net output of P and K (Fig. 11.7; Olde Venterink et al. 2002). At highly productive sites, hay-harvesting resulted in net output of N, P, and K (Koerselman et al. 1990). When the annual nutrient output was expressed as percentages of the total soil pools, it was found that net output of K (1–20 % of soil K pool) was mostly much larger than that of P (generally 0.5–3.0 %) or N (0–3 %). Hence, in the long-term, hay-harvesting seems to induce K limitation, particularly when they are drained for a long time (Van Duren et al. 1981; Van Duren and Van Andel 1997).

11.3.3 Internal Eutrophication

More intensive agricultural use not only leads to lowered water tables but also to increased concentrations of NO_3^- and SO_4^{2-} in ground- and surface water. Effects of water table alterations receive much attention, but there is confusion about its consequences for nutrient kinetics. Desiccation often leads to eutrophication in peatlands (Grootjans et al. 1996). However, this is not always

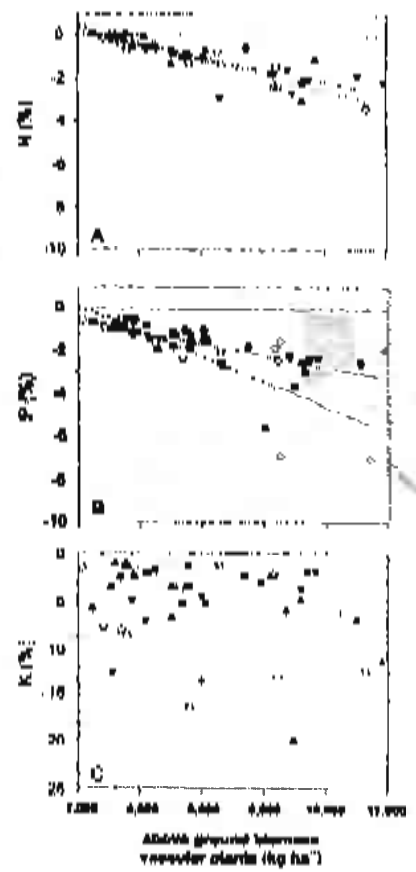

Fig. 11.7 Net annual output of N, P and K by hay-harvesting as a percentage of the total N, P, and K pools in the soil (0–10 cm), in relation to the above-ground biomass of vascular plants in fens (*open circles*) and meadows (*filled circles*) (Olde Venterink et al. 2002; reproduced by permission of the Ecological Society of America)

true. fens that are limited by P and K, in particular, usually respond to drainage with a decreased productivity. This had already been noticed by Russian researchers in the early 1930s during the communist period, when extensive mires were reclaimed for agriculture. They found a marked decrease in groundwater-fed fens after drainage (Ozushko 1934, cited in P'yavchenko 1976). Ozushko warned: "That money invested in drainage, is wasted unless the drained area is then used for the cultivation of agricultural crops". At that time, Russian agro-scientists already knew that P and K were limiting in many drained mires and not N.

Restoring desiccated fens is not simply a matter of increasing water levels. Recent research shows that rewetting with sulfate-rich surface water, in par

ticular, may lead to increased eutrophication in fens. High SO_4^{2-} availability can induce PO_4^{3-} mobilization in wetland soils. This may occur when the former discharge of iron-rich groundwater is not restored. In such a case, the groundwater is replaced by sulfate-rich surface water. When the water stagnates, all oxygen is consumed and sulfate reduction is stimulated when the outside temperatures are sufficiently high. Sulfate reduction produces sulfides, which in the presence of iron form FeS_2 (pyrite). PO_4^{3-} bound to iron hydroxides is then released to the pore water or the stagnating water body (Smolders et al. 1995; Lamers et al. 1998).

Although the active manipulation of the internal mobilization of nutrients has been practiced since medieval times by farmers trying to improve their crop yield by regular flooding with surface water, present biogeochemical research in peatlands often overlooks the alterations in nutrient mobilization rates due to such anthropogenic water quality changes. Flooding with surface water during the summer, in particular, can be very harmful for species-rich fens and may result in the rapid growth of tall grasses, such as *Phalaris arundinacea* or *Glyceria maxima*. Periodic local desiccation of the sediment during the summer, which is a natural phenomenon in fens, is important in sulfate-rich wetlands because this determines the ability of the soil to retain phosphates. The oxidation of FeS_2 again increases the reactive Fe^{3+} concentration, stimulating P-binding in the top layer (Lucassen et al. 2004a).

The role of increased NO_3^- levels in the groundwater due to intensive agricultural fertilization is complex. Nitrate may increase the redox potential but also leads to a higher binding capacity of P due to FeS oxidation. A high input of nitrate inhibits SO_4^{2-} reduction because it is a more favorable electron acceptor and also the oxidation of FeS_2 will increase the sulfate levels. So, high nitrate levels in the groundwater may counteract the negative effects of high sulfate concentrations. On the one hand, NO_3^- leaching into the groundwater increases the risk of eutrophication of groundwater discharge areas by mobilizing SO_4^{2-}, in FeS_2-containing soils or aquifers. On the other hand, high concentrations of NO_3^- may prevent P eutrophication (Lucassen et al. 2004b). As a rule of thumb, to avoid eutrophication due to internal release of nutrient in fens, one should not increase surface water tables above the potential groundwater table.

11.4 Seed Bank and Seed Dispersal

An increase in species richness over time following restoration depends on the environmental site conditions and on seed availability, germination, and survival. Fine-tuning of the environmental conditions for restoration is already a difficult task, but very often seed availability and also seed dispersal are restricting restoration prospects considerably (Bakker et al. 1996).

When degeneration processes have been taken place for a long time, seed banks might have been depleted and ways to introduction of locally extinct species have to be considered. Sometimes lack of seed availability can be improved by stimulating natural dispersal processes, such as flooding (Middleton 1999).

11.4.1 Seed Banks

Wet grassland species have a limited ability to regenerate by seed, because these species usually have only transient seed banks (Bekker et al. 1997, 2000) and seeds of rare and endangered species also have a comparatively low persistence (Fischer and Stöcklin 1999). In addition, seed dispersal distances are usually low (Bullock et al. 2002; Soons 2004) and thus it can be questioned whether seed persistence and dispersal significantly contribute to the restoration of species-rich wet grassland communities.

Jensen (1998) analyzed seed banks and the local seed rain of abandoned wet grasslands and found that the (anemochorous) seed rain of the investigated sites was indeed dominated by species of the aboveground vegetation, implying that long-distance seed dispersal by wind is a rather rare event in this habitat type. In contrast, the seed bank of late successional stages, in which most typical wet grassland species had disappeared in the vegetation, always contained some elements of the typical wet grassland flora. These results at least partly contradict the hypotheses of transient seed banks of most grassland species. Jensen's research group, therefore, carried out seed burial experiments to examine directly the seed mortality and longevity of approximately 45 species of the regional fen flora (Schütz 1997, 1998, 1999; Jensen 2004a, b; Brändel 2004a, b). These studies revealed that the mortality of artificially buried seeds of most species was low, that only a few species exhibited transient seed banks, and that the seeds of almost half of the investigated species were able to persist in the soil for at least five years (i.e., these seed banks were long-term and persistent). To explain such differences in seed longevity estimates, Jensen (2004b) compared the seed-longevity estimates for a set of 230 herbaceous wetland species from Northern Germany using different methods of seed bank analysis (e.g., burial experiments vs classic soil seed bank sampling) and various (indirect) indicators for seed bank classification (presence/absence in vegetation and seed bank, depth distribution in the soil; see Thompson et al. 1997). This analysis showed that burial experiments generally resulted in the highest longevity estimates. The criterion stated as "presence in the vegetation, but absence in the seed bank" is the criterion used by Thompson et al. (1997) to classify a species as "transient". However, many species classified as transients in this way had unrealistically low longevity estimates. From these results, it can be concluded that many typical wet grassland species are able to persist at least some years as viable

seeds in the soil. We may conclude that much more experimental evidence is needed to develop realistic criteria for estimating seed longevity of soil seed banks.

11.4.7 Seed Dispersal

Water is an important dispersal vector and a number of studies have quantified hydrochorous seed dispersal in wetlands, such as clear streams in an alpine zone (Bill et al. 1999), forested temperate wetlands (Middleton 2000), boreal rivers (Andersson and Nilsson 2002), and a small lowland river (Boedeltje et al. 2003). All these studies revealed that running water might disperse seeds over quite long distances. Vogt et al. (2004) analyzed the hydrochorous seed dispersal in a fen in Northern Germany (Elder Valley) by means of seed traps. This two-year study revealed very large temporal and spatial variation in both quantity (seed number) and quality (species composition) of the hydrochorous seed transport. Overall, seeds of almost 200 different taxa were found in the traps, which is over 60 % of the regional species pool. Seed transport was dominated by (common) species which were abundant along the river shores, but rare and regionally endangered species were also present, albeit only in low quantities. Wind dispersal may also be important in these wetlands and human effects on landscapes, including habitat fragmentation and eutrophication, may affect the colonization capacity of species (Soons et al. 2004).

11.5 Fen Restoration: An Example From Hungary

11.5.1 Introduction

Restoration provides a way to offset some habitat loss for the rare species of fens and other wetland habitats (Middleton 1999; Grootjans et al. 2002; Lamers et al. 2002; Wheeler et al. 2002), particularly because more than 50 % of the world's wetlands have been lost (Munro 1988). Few specific estimates of loss for fens have been made in North America, but in some regions of the United States fen loss exceeds 40 % (Bedford and Godwin 2003). In Europe, less than 1 % of the original mire area has remained in most countries (Joosten and Clarke 2002), although some countries have some floodplain wetlands that are nearly natural (e.g., Turkey, Romania, France; Wenger et al. 1990). In eastern European countries, some 10–50 % of the former mire area still remains. In some Scandinavian countries, European Russia, and Romania, the mires are the best preserved (>50 % of the mires remaining). On a

world scale, mire losses are highest in Europe due to its high population pressure on nature and the climatic suitability for agriculture and forestry. In Hungary, in particular, wetland losses are large. It is estimated that 97 % of the natural wetlands have been drained and converted to agricultural fields (Lajer 1998); and many of the remaining wetlands are damaged. While the restoration of fens may be a way to offset these worldwide losses of fens and related biodiversity loss, few examples exist of successful fen restoration, so this example from Hungary is of interest to our discussion.

11.5.2 Destruction and Restoration of a Fen System in Hungary

Fen restoration is less commonly attempted than some other types of restoration, possibly because of the difficulties of reestablishing the hydrology associated with groundwater. One attempt to restore fens has been made in the Hanság, a wetland that was once the largest fen system in the Carpathian basin of Hungary (55 000 ha; Szenkendi 1938). Before the nineteenth century, several rivers from the Alps flowed into the Danube valley, disappeared into the huge floating fen system, and re-appeared at the outflow. Local people used the biological resources of the fen for fishing, hunting, collecting eggs of water birds, hay-making, and reed-harvesting. However, human settlements and agricultural fields near the fen were continuously threatened by inundation, so attempts were made to drain the Hanság, beginning in the nineteenth century. Despite drainage and disturbance, the Hanság retained large patches of natural vegetation in the wet meadows and forests; and the vegetation of aquatic communities survived in drainage channels. Because of its natural value, the Hanság was designated a protected natural area in 1976 and in 1994 as the Fertő–Hanság National Park. It is within this park that the restoration project was conducted.

To restore the wetlands, managers of the Fertő–Hanság National Park reflooded 400 ha of poor quality grassland, with the support of the Hungarian and Dutch governments. To re-engineer the restoration sites, dikes were built around the three separate wetlands. Gravity transported water through sluices from the river Rábca and the channel Klamezés (Fig. 11.8). The first and second units were flooded in spring 2001 and the third unit in autumn 2001. The water-level has stayed constant at 0–100 cm above the soil surface. While a floating fen was not restored by this procedure, a wetland with an open-water habitat was created for water-birds and fen plants. Invasive plant species such as *Solidago gigantea* were reduced in the restored wetland.

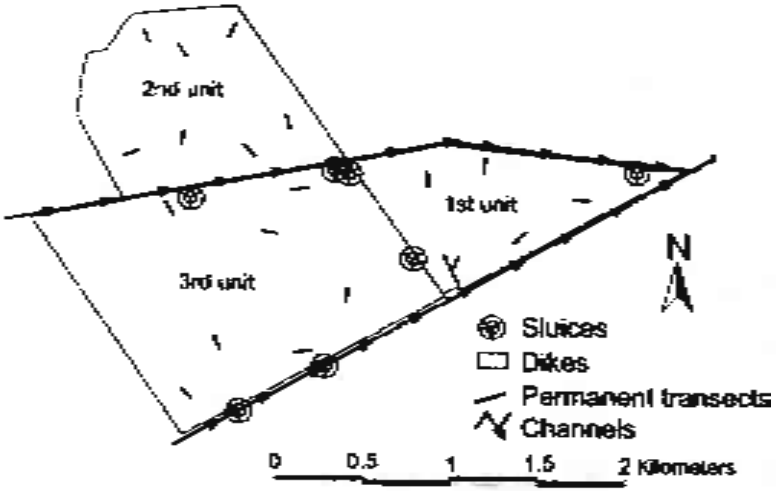

Fig. 11.8 Map of the restoration area, showing the location of permanent transects, in the Hanság fens of the Fertő-Hanság National Park, Hungary (Margóczi, unpublished data)

11.3.3 Monitoring and Evaluation of the Created Wetland

Monitoring of the restoration site was conducted in two ways, including field sampling and remote sensing. Field surveys of the vegetation were conducted by recording percent cover within 5 × 5 m phytosociological releves along 21 permanent transects, 100 m long each, during 2001–2003. To create maps of vegetation change over time, fine resolution aerial orthophotos were compared every third year following the hydrologic restoration and a GIS database was developed. A rough vegetation map was made before the flooding began. The main vegetation types before flooding included fen meadows with large sedges (57 % *Carex riparia*, *C. acutiformis*), wet meadows (28 % *Alopecurus pratensis*, *Festuca arundinacea*), and reeds (5 % *Phragmites australis*, *Glyceria maxima*; Takács and Margóczi 2002). Important shifts in vegetation type were apparent during the first three years of flooding (Fig. 11.9). During the second year of flooding, upland vegetation disappeared; the area of the *Typha*, aquatic plants, and open water increased, while the area dominated by sedges decreased.

Natural wetland communities have redeveloped in the Fertő-Hanság National Park and are an attractive breeding and feeding area for birds. However, the hydrology is considerably different from the original fen, so the natural plant communities undoubtedly differ from those communities present

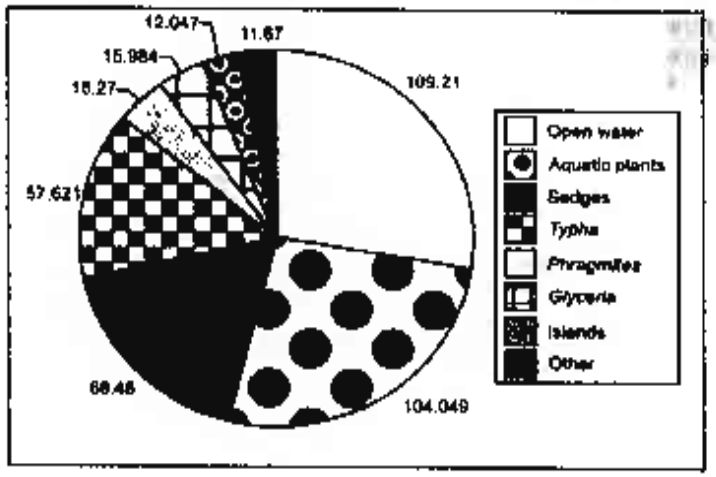

Fig. 11.9 Area (ha) of the main vegetation types in the flooded units (Units 1–3) in the Hanság fens, Hungary, in 2003. Area sizes were computed using raster statistics from the GIS database (Margóczi, unpublished data)

before the drainage of the ancient fen. While fen vegetation is not expected to re-develop, the Hanság National Park will continue to monitor the wetland restoration. Among the challenges that the managers of this national park face is that the restoration site is surrounded by agricultural lands, so that a considerable amount of water management and soil conservation in the regional landscape may be necessary to save the natural values of these wetlands.

11.6 Concluding Remarks

In cases without much human influence, fen development was remarkably stable and vegetation types existed for centuries until they were replaced by meadows and alder wood. Nowadays fens cannot be maintained without management such as mowing or grazing. Nevertheless biochemical mechanisms exist that can retard a rapid succession to shrubs and forest. Such mechanisms are governed by subtle hydrological processes that keep the fen in a good condition.

A prerequisite for successful nature management is the creation of suitable habitat conditions (ancient status, hydrology). There will be no adequate fen restoration without a proper assessment of the hydrological functioning of

the fen system. It is very important that stagnation of surface water in the summer is prevented in restoration projects dealing with desiccated fens, in particular when the groundwater or surface water is rich in sulfate. The best option, which is not always available, is to stimulate the discharge of unpolluted (sulfate- and nitrate-poor) groundwater in such a way that this groundwater can flow through the system, leaving iron behind to contain the phosphates in the top soil.

Management measures are needed to counterbalance, for instance, the increased annual nitrogen input from atmospheric deposition. This may be accomplished by annual hay-harvesting, but it should be realized that hay-harvesting may induce a shift in the type of nutrient limitation, which also may affect biodiversity (Olde Venterink et al. 2003).

From studies on soil seed banks and seed dispersal, we may conclude that, until now, both seed persistence in the soil and dispersal distances of wet grassland species have been rather underestimated. Both types of seed sources can be manipulated by nature management, both in conservation and restoration projects. For successful restoration, both temporal (time since alteration of the habitat) and spatial (degree of fragmentation/isolation) aspects have to be taken into account.

We may expect that, in the near future, large areas on low-lying peat soils in large part of Europe and North America can no longer be maintained as intensely used agricultural areas. This offers good opportunities for the development of eutrophic marshes, as in our example of the Hungarian wetland restoration. The reflooding of former agricultural areas usually results in very eutrophic soil conditions (Richert et al. 2000), in particular when the surface water is rich in sulfates and the soil has been iron-depleted due to long-term drainage. Under such conditions persistent marsh species, such as *Glyceria maxima*, *Typha* spp. and *Phalaris arundinacea* can expand rapidly, but these species do not usually form peat, although they can rapidly form highly productive marsh vegetation, suitable to sustain large populations of waterfowl. To restore the peat-forming function of the mire, the vegetation should consist of marsh species with less easily degradable tissue (*Phragmites australis*, tall *Carex* spp; Richert et al. 2000). Restoring all of the fen qualities and functions will not be possible in many cases where restoring the hydrological system is not yet an option. Setting clear targets which can meet the opportunities presented, combined with a realistic strategy for future possibilities, may prevent much disappointment. A well explained modest result sometimes does more good for the public support than a scientifically sound project that lacks community or administrative support.

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12 Social Learning in Wetland Development

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12.1 Introduction

An important issue in wetland planning and design is the most appropriate relationship between expert knowledge and public participation. How is it possible to organise a meaningful input from the public and what does this mean for the role of specialists? The insight that stakeholders, for instance local farmers, must be involved in the development of wetlands is widely accepted. But the extent of this involvement appears to be a matter for discussion. Some argue that only sound research can be the basis for the analysis of the actual situation and for setting objectives in planning and design. They claim that the most important intention of wetland development is to improve their ecological value. A consequence of this paradigm is that the involvement of stakeholders would be limited to including them in a 'value-free' planning process that is to lead to achieving a given objective. At most, stakeholders might be consulted in order to check whether data are well interpreted and processed; and they may eventually be given a role in wetland management.

An alternative to this specialist-dominated planning approach is social learning. In rural resource management, social learning "reflects the idea that the shared learning of interdependent stakeholders is a key mechanism for arriving at more desirable futures" (Leeuwis and Pyburn 2002, p. 11). Röling puts it as follows: "Social learning is about the interactive way of getting things done in theatres with actors who are interdependent with respect to some contested natural resource or ecological service. The interactive way of getting things done is based on conflict resolution, negotiated agreement, shared learning, convergence of goals, theories, and systems of monitoring, and concerted action. It stands in sharp contrast to instrumental use of technologies to control nature for assumed human purposes. It also stands in sharp contrast to economics which ascribes people with reasons (in assuming

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EXHIBIT 7