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Forest to Reclaimed Mine Land Use Change Leads to Altered Ecosystem Structure and Function

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The United States' use of coal results in many environmental alterations. In the Appalachian coal belt region, one widespread alteration is conversion of forest to reclaimed mineland. The goal of this study was to quantify the changes to ecosystem structure and function associated with a conversion from forest to reclaimed mine grassland by comparing a small watershed containing a 15-year-old reclaimed mine with a forested, reference watershed in western Maryland. Major differences were apparent between the two watersheds in terms of biogeochemistry. Total C, N, and P pools were all substantially lower at the mined site, mainly due to the removal of woody biomass but also, in the case of P, to reductions in soil pools. Mineral soil C, N, and P pools were 96%, 79%, and 69% of native soils, respectively. Although annual runoff from the watersheds was similar, the mined watershed exhibited taller, narrower storm peaks as a result of a higher soil bulk density and decreased infiltration rates. Stream export of N was much lower in the mined watershed due to lower net nitrification rates and nitrate concentrations in soil. However, stream export of sediment and P and summer stream temperature were much higher. Stream leaf decomposition was reduced and macroinvertebrate community structure was altered as a result of these changes to the stream environment. This land use change leads to substantial, long-term changes in ecosystem capital and function.

Keywords: biogeochemistry; carbon; coal mining; forest; Georges Creek basin; hydrology; land-use change; macroinvertebrates; mine soils; nitrogen; phosphorus; reclaimed mine

Introduction

One of the major land use changes in the past several decades in the Appalachian region of the United States is conversion of forest to reclaimed mine lands (Loveland et al. 2003). For example, in Georges Creek watershed in Maryland, mined area increased from 3.8% in 1962 to 15.5% in 1997 (Negley 2002). In 1998 in southern West Virginia, active and reclaimed mine acreage ranged from 5.4 to 10.6% of the land area among eight river basins, which represented a 42% increase from 1994 (West Virginia Department of Environmental Protection 2004). In northern and central Appalachia (Pennsylvania, Maryland, West Virginia, Virginia, Tennessee, and Kentucky) over 2.6 million acres (1.1 million ha) of land are listed as disturbed by active mining operations (Office of Surface Mining 2004). Coal mining is expected to continue to expand throughout this century as the demand for electricity and alternative fuels (e.g., coal gasification) grows. Therefore, we can expect continued conversions of forest to reclaimed mineland throughout the eastern coal belt.

Surface mining typically occurs in three stages. In the first stage the site is cleared of vegetation and the uppermost soil horizons are removed and stored leading to homogenization of the material. The second stage consists of removal of soil and rock overburden, extraction of the coal, and replacement of the homogenized overburden to approximate original contour. The third stage is reclamation or reestablishment of vegetative cover. The homogenized soil is replaced, graded, and seeded. The most common post-mining land uses are hay land and pasture, and the productivities of these reclaimed grasslands can be quite low because the constructed soil is often a poor medium for plant growth.

A great deal of research effort has been invested in understanding the soil factors that lead to poor reclamation success, such as nutrient scarcity (Plass and Vogel 1973, Roberts et al. 1988, Vetterlein et al. 1999), low organic matter content (Boerner et al. 1998, Akala and Lal 2000, 2001), low pH (Thurman and Sencindiver 1986, Roberts et al. 1988, Johnson and Skousen 1995), high bulk density (Johnson and Skousen 1995, Burger and Zipper 2002), and poor soil texture and structure (Thurman and Sencindiver 1986, Roberts et al. 1988). However, it is not clear if and when ecosystem processes like nutrient cycles are being restored in these reclaimed systems because there is a lack of ecosystem-level studies and biogeochemical information, particularly on the links between the terrestrial and aquatic systems.

With a conversion from forest to reclaimed mineland comes major changes in vegetation community, wildlife habitat, and soil structure and properties (Johnson and Skousen 1995, Williams et al. 1995, Boerner et al. 1998, Simmons et al. 2005). We know from research on land-use change that these changes will lead to major changes in biogeochemical cycles, hydrology, stream physicochemical characteristics, and fauna; however, the degree and direction of these changes is unknown. Landscape ecologists assert that quantitative assessment of the full range of ecosystem responses to a land-use change—both intended and unintended responses— is needed to make fully informed decisions about land use (DeFries et al. 2004). Currently, we do not have the ability to predict ecosystem responses to mining, much less understand the mechanisms behind these changes. Although the amount of land area that undergoes this conversion annually is a relatively small fraction of total land area, the magnitude of the change is large and could potentially result in significant modifications of ecosystem processes at the regional scale. Cumulative landscape conversions, like urbanization, have been shown to have large scale impacts (Boward et al. 1999). Therefore we decided to conduct an ecosystem-scale study of a mined and reclaimed small watershed.

Compared to studies of other ecosystem types, designing a typical paired watershed experiment that includes a mined and undisturbed watershed with several years of pre-disturbance calibration data is fraught with difficulties. For example, the mining disturbance is not a point disturbance like logging, but is a process that may take as long as 10 years; much longer than most funding cycles. Furthermore, maintaining permanent sampling plots and stream gages within an active strip mine where the soil and bedrock are removed is generally not possible. Another challenge is that truly undisturbed watersheds in the eastern United States are uncommon. Finally, most mines in the eastern United States are located on private land, which means that gaining access can be problematic. With these limitations in mind, we settled on a design in which we compared two watersheds with similar initial (pre-disturbance) site characteristics that are recovering from two very different types of disturbance.

Our main objective was to quantify in a holistic manner the differences in ecosystem structure and function between a watershed subjected to mining and a watershed subjected to a relatively minor logging disturbance at approximately the same time. This is the first study that quantifies the differences in ecosystem-level, biogeochemical process between a forested and mined ecosystem. It is a first step toward a better understanding of the effects of coal mining at the regional scale.

Methods

Site description

The two study watersheds were located on Dans Mountain (39°35′ N, 78°54′ W), just south of Frostburg in western Maryland, USA within the Georges Creek basin. The watersheds were selected after an intensive basin-wide search for adjacent or nearly adjacent paired watersheds under 50 ha in size using the following criteria: slope, aspect, elevation, time since disturbance, historical vegetative cover, accessibility/ownership, and soil type. The two watersheds that were chosen were the most similar to each other and the most representative of Georges Creek basin (Table 1). The two study watersheds have slopes close to the mean for Georges Creek basin (10%) and are on a soil type that is abundant in the basin.

Table 1.

Characteristics of the three study sites.

NEF watershed is a 3.0-ha subwatershed of Neff Run watershed covered with a predominantly 19-year-old (at the beginning of the study) mixed hardwood forest with some coniferous trees. Most of the subwatershed was selectively logged in 1980–1981 according to tree ring data and aerial photos (K. Kuers, *unpublished data*). NEF contains a first-order tributary to Neff Run (TNEF1). More detailed descriptions can be found in Castro et al. (2007) who studied the nitrogen cycle of NEF and characterized the watershed as N saturated. Although we cannot claim that NEF is undisturbed, we consider it a reference watershed because it was disturbed at approximately the same time as the treatment watershed, it has not been disturbed since, and the disturbance (selective logging) was relatively minor compared to mining and reclamation. Furthermore, our goal is to contrast two land-use types, and the vast majority of forests comprising the forest land-use category in the mid-Atlantic region are second-growth forests.

MAT is a 27.2-ha subwatershed of Matthews Creek watershed located ∼500 m from NEF and at a slightly higher elevation (Fig. 1). Prior to 1981, MAT was similarly covered with a predominantly mixed hardwood forest with some conifers but was clear-cut in 1981 according to aerial photographs, tree ring data, and mining permit records. Approximately one-half the area of MAT (12.5 ha) was surface mined for coal from 1982 to 1985. Reclamation, which comprised reapplying ∼30 cm of topsoil that was scraped from the site after clear-cutting, amending with fertilizer, and seeding with a mixture of nonnative pasture grasses, was completed in 1986. Since that time it has been managed as an ungrazed, periodically mowed grassland. We refer to the reclaimed mine portion of the watershed as MAT-R. The 14.7 ha that were mostly clear-cut but not mined (designated as MAT-F) were allowed to regenerate as forest, resulting in a community consisting of more oak and less sugar maple and black cherry than NEF. TMAT1 is a zero-order, intermittent stream that flows through a constructed, riprap channel. TMAT1 is located entirely in MAT-R; there are no established channels in MAT-F. Soil characteristics of both sites are described in Simmons and Currie (2005).

Fig. 1.

Topographic map showing the relative locations of the watersheds of the study area in western Maryland, USA. TNEF1 is the stream that drains NEF watershed. TMAT1 is the stream that drains MAT watershed. TMAT1 is a tributary of Matthews Run. MAT is subdivided into a reclaimed mine section (MAT-R) and a forested section (MAT-F).

Soils

Within both NEF watershed and the MAT-R site, three randomly located and permanent 100-m transects were established at orientations of 0°, 120°, and 240° from north. Soil samples were collected from each 3-m segment along the length of each transect in August 1999 in NEF and in August 2000 in MAT-R for a total of 33 samples. Samples from two horizons were collected quantitatively using a 10 × 10 cm template and two 5 cm diameter bulk density cores to a depth of 10 cm for the Oe/Oa and mineral soil horizons. In NEF, the Oi horizon was collected quantitatively using a 10 × 10 cm template. Bulk density was determined before sieving with a 2-mm mesh sieve. Fine roots were manually sorted from the core samples during the sieving process, dried, and stored for tissue C and N analysis as described in *Methods: Vegetation*. Subsequent tests were conducted only with the <2 mm fraction of soil. Sieved soils were analyzed for pH (0.01 $CaCl_2$), total C and N by CHN Analyzer (Carlo-Erba, Milan, Italy), and total P by dry ashing (Olsen and Sommers 1982). Thickness of the Oe/Oa horizon was determined at 1-m intervals along each transect. In 2001, a single transect was established in MAT-F, and it was sampled in the same manner as the other transects.

Three 20 × 20 m randomly located permanent sampling plots were established in NEF and MAT-R in 1999 and each subdivided into 16 5 × 5 m subplots. Soil temperature at 5 cm depth was measured using thermistors with dataloggers (Onset Computer, Pocasset, Massachusetts, USA) in one subplot within each plot. During the 2000 growing season, monthly in situ rates of net N mineralization and net nitrification were measured in three randomly selected subplots in each of the three permanent plots using the buried bag technique for a total of nine replicates per four week sampling period. During each monthly sampling period, adjacent pairs of 5.4 cm diameter cores were collected to a depth of 10 cm of mineral soil. These cores were separated into organic and mineral horizons for incubation and analysis (Aber et al. 1993).

Fungal species were determined from triplicate soil cores taken from each permanent plot on three dates during the growing season from 2001 to 2004. Subsamples (1 g) of each core were placed into sterile distilled water, vortexed, and diluted by factors of 10^2 and 10^3. Subsamples were incubated on Rose Bengal agar plates (Sigma-Aldrich, Milwaukee, Wisconsin, USA) where morphotypes were identified and colonies were counted.

In order to evaluate the functional diversity of the microbial community (i.e., carbon substrate selectivity) additional subsamples were plated onto BIOLOG EcoPlates (Hayward, California, USA). At 24, 48, 72, and 96 hours, readings of mean well color development were made using the BIOLOG Microlog 3E system. Use of a particular carbon source is indicated by tetrazolium chloride color change induced by cellular respiration.

Soil parameters were subjected to a nested ANOVA with site (NEF, MAT-R, MAT-F), horizon, and transect (or plot) as treatments followed by Student-Neuman-Keuls (SNK) test to distinguish among treatments. For variables measured in only one horizon, data were pooled by transect (or plot), and a one-way ANOVA was used to determine site differences. Soil temperatures were compared using repeated-measures ANOVA with date and site as treatments. All alpha values were set at 0.05 unless noted otherwise.

Vegetation

Woody vegetation >1.3 m tall was characterized using circular inventory plots (40-m^2 each) distributed in a grid across each forested site (19 plots in NEF and 26 in MAT-F). Species and dbh were recorded for each tallied tree. Aboveground woody biomass of the trees was calculated using whole-tree allometric equations developed by Brenneman et al. (1978) for a similar forest type in West Virginia. Individual equations were used for tree species for which they were available, and the equation for a tree of similar specific gravity was used for the remaining species. Small branches (<1 cm) were assumed to be 12.5% of the total aboveground tree biomass, based upon data collected by Adams et al. (1995) on a similarly aged stand in West Virginia.

In July 2003, foliage, bole, and small branch samples were obtained at each watershed from three species (chestnut oak, red maple, and northern red oak). These species represented the species with the highest, lowest, and mid-range foliar N concentrations. Three trees from each of three size classes (0–5 cm, 5.1–15 cm, and >15 cm) were sampled for each species (nine samples for each species for each tissue type). Foliar nutrient pools were calculated using either litterfall mass (NEF) or allometric equations (MAT-F) and weighted-average tissue-nutrient concentrations.

In the mined portion of MAT watershed (MAT-R), which was entirely grassland with only occasional small trees, living and dead biomass was measured on three dates in 20 0.1-m^2 quadrats per plot by clipping. Biomass was sorted into current-year dead, past-year dead, grasses, sedges/rushes, legumes, forbs, and mosses/lichens.

Tissue samples were dried to 70°C, ground in a Wiley mill, and then ball-ground before being analyzed for total N and total C using a CHN analyzer (NC-2100; Carlo-Erba Instruments). Subsamples were later dried to 105°C and all N and C values were corrected from 70° to 105°C (typically less than a 1% difference).

Five litterfall collectors (0.23 m^2) were each randomly assigned to a subplot within each permanent sampling plot from 1999 through 2002. Litterfall was collected monthly, dried, weighed, composited by date, and processed as previously described for plant tissue.

The General Linear Model (SPSS, Chicago, Illinois, USA) was used to test for significant effects of species and site on tissue N and C concentrations. Tukey's hsd test was used to separate means for variables determined to be significant $P = 0.05$).

Precipitation and hydrology

Stream stage and discharge in TNEF1 and TMAT1 were measured using stream gages installed prior to the commencement of the 2000 water year on 1 October 1999. Because both sites lacked natural bedrock controls, precalibrated “Montana” (i.e., truncated Parshall) flume was installed at each site. Each flume (manufactured by Free Flow, Omaha, Nebraska, USA) was constructed of fiberglass and anchored to wooden wingwalls buried in the streambed and bank. Analog measurements of stage were made continuously using a float and counterweight design. All stage records were digitized and subsequently converted to hourly stage and discharge using rating curves obtained from the flume manufacturer. Analog recorders were replaced with digital (Unidata Model 6541; Lake Oswego, Oregon, USA) recorders that provided hourly data during the second year of the project. Instantaneous stream discharge data were aggregated for computation of mean daily and annual values and normalized by the respective watershed area to provide comparable runoff values in consistent units of millimeters.

Hourly precipitation was measured using two weighing-type rain/snowfall gages (Belfort Instrument, Baltimore, Maryland, USA) located in MAT-R (Fig. 1). Rain gages operated on 8-day hourly charts and were sensitive to precipitation depths greater than 1 mm. Data obtained from a National Weather Service cooperative observing station in Frostburg, Maryland (located ~5 km north of the site) provided data for the period prior to 18 December 1999 when the gages were inoperative.

Grab samples of stream water were collected at irregular intervals during the course of the project and analyzed for several parameters. Periodic sampling was attempted in the first two years, but the intermittent nature of the two streams made this impractical. Subsequently, water sampling was timed to correspond to periods when the streams were likely to be flowing. Stream and precipitation samples were analyzed for nitrate-N using suppressed ion chromatography (Dionex DX500 instrument, Sunnyvale, California, USA), ammonium-N using continuous-flow colorimetry, DOC using coulometry, and total suspended solids (TSS) by filtration. Monthly and annual stream fluxes were computed assuming that daily concentrations could be linearly interpolated from sequential samples (Eshleman 2000). Annual rates of NH_4^+ and NO_3^- deposition were estimated by multiplying the total annual precipitation amount by the annual volume-weighted mean concentrations of NH_4^+ and NO_3^-.

Stream ecology

Leaf packs for incubation in streams were made with five grams of dried red maple (*Acer rubrum*) leaves collected from a single source and nylon mesh bags (1-cm mesh). Twelve leaf packs were deployed in each of the following locations: near the flume in TNEF1, near the flume in TMAT1, upper Matthews Run ~0.5 km upstream from the confluence with TMAT1, at the confluence of TMAT1 and Matthews Run.

Four leaf packs were harvested from each sampling location at 1–4 month intervals (depending on the time of year), placed in plastic bags, and transported on ice to the laboratory. This sampling regime was replicated a total of three times between July 2001 and November 2002. Harvested leaves were rinsed over a screen to remove sediment and to collect macroinvertebrates. Leaves were dried at 70°C for 48 hours to determine mass loss. The decomposition rate constant, *k*, was calculated as the slope of the curve of ln(percentage of mass remaining) vs. time. Macroinvertebrates from the leaf packs were sorted, counted, and identified to family. Counts from all sampling dates were combined and metrics calculated for each site (Barbour et al. 1999).

Results

Soil

The soil of the reclaimed mine site, MAT-R, was very different in nature from the soil in the forested areas. Structurally, MAT-R was covered by a thin Oe/Oa horizon, which was discontinuous in some areas, overlying a thick A horizon (Table 2). Because the soil was so young (15 years), horizon development was minimal. In contrast, the forested sites had a distinct Oe/Oa horizon, ~6 cm in thickness, overlying several well-developed mineral soil horizons. The mineral soil at MAT-R was denser, had a higher pH and lower C, N, and total P concentrations than the two forested mineral soils although the C:N ratio was intermediate between the two. The high bulk density is likely a result of compaction by heavy equipment during regrading of the site. The higher pH could be attributed to the relative lack of two acidifying processes at MAT-R: nitrification (described in *Results: Nitrogen pools and fluxes*) and production of organic acids by leaf decomposition. Low C and N concentrations are typical of minesoils and result from accelerated decomposition and leaching from soil handling during mining (Davies et al. 1995).

Table 2.

Soil characteristics of the three study sites.

The infiltration rate at MAT-R (<3 mm/h) was orders of magnitude lower than at NEF (300 mm/h), probably as a result of the higher bulk density. The low infiltration rate means that even a moderate rainfall intensity could lead to inundation and surface runoff at the former mine site. Mean annual soil temperature at 5 cm was very similar at both sites; however, the mean daily range (the difference between the daily maximum and daily minimum temperature) was significantly greater at MAT-R than at NEF, meaning that daily fluctuations were more extreme (Table 3). Furthermore, daily maximum soil temperatures in the summer months frequently exceeded 30°C at MAT-R whereas at NEF they never exceeded 25°C. Consistent with the observation of lower infiltration and warmer summer soil temperatures was the finding that gravimetric soil moisture during the growing season was consistently and significantly lower at MAT-R than at NEF (Table 3).

Table 3.

Soil temperatures in forested (NEF) and reclaimed mine (MAT-R) watersheds.

The number of fungal colony-forming units was significantly lower in the mined site than in NEF, indicating a lower potential activity of fungi. This could be a result of the more extreme and variable temperature and moisture conditions or the lower C and N concentrations in MAT-R. The species composition also varied considerably. More transient taxa such as the *Phycomycetes* and *Mucor*, were apparent in the mined site. These genera, as indicated by Stolp (1988), are often thought of as first colonizers and seem to colonize areas first where free or labile carbohydrates are present. Although present in both, secondary colonizers, such as *Penicillium*, *Aspergillus*, and *Trichoderma*, were more abundant in the forested site. Zak (1992) indicated variations in microbial community in reclaimed mined sites over time as colonization occurs. Fresquez et al. (1986) found that population sizes of several groups of microorganisms were similar in mine spoil soils and undisturbed soils after four years, indicating that recolonization can be rapid.

Utilization of carbon sources varied by date, but the microbial community (fungi plus bacteria) in MAT-R always used a greater variety of substrates than the NEF community (Table 4). The temporal variation appeared to be moisture dependent but also may have been a result of nitrogen availability as indicated by Dix and Webster (1995). Thus, compared to NEF, the microbial community at MAT-R could be characterized as consisting of more generalist species, as would be expected in a relatively less stable soil system early in succession.

Table 4.

Number of fungal colony-forming units and number of carbon sources utilized in soil cores taken from NEF and MAT-R.

Carbon pools and fluxes

As one would expect when comparing a forest ecosystem to a grassland ecosystem, aboveground C pools at MAT-R were <5% the size of the pools at NEF and MAT-F (1200 vs. 52 000 and 47 000 kg C/ha, respectively; Table 5). The main difference was the lack of large woody biomass pools in the reclaimed mine site. In contrast, belowground C pools were much more uniform owing to the very similar mineral soil C pools. Belowground C pools were estimated to be 23 900, 32 900, and 62 600 kg C/ha for MAT-R, NEF, and MAT-F, respectively. Thus, overall NEF and MAT-F contained substantially larger C pools than MAT-R, especially when one considers that coarse woody debris and coarse roots, which would be virtually nonexistent at MAT-R, were not included in the estimates.

Table 5.

C budgets for TNEF, MAT-F, and MAT-R watersheds.

Nitrogen pools and fluxes

Nitrogen pools followed a similar pattern to the carbon pools. The total N pool was much larger at NEF (>2650 kg $N \cdot ha^{-1} \cdot yr^{-1}$) and MAT-F (>3328 kg $N \cdot ha^{-1} \cdot yr^{-1}$) compared to MAT-R (1455 kg $N \cdot ha^{-1} \cdot yr^{-1}$; Table 6). There are three main reasons for this: (1) large amounts of N in woody biomass in the forests that MAT-R lacked, (2) much thicker Oe/Oa pools in forested sites, and (3) generally higher concentrations of N in foliage, litter, roots, and soil. For example, the overall mean fine-root N concentration in NEF was 2.06 vs. 1.23% in MAT-R. In addition, mean species N concentration in foliage ranged from 2.47 to 3.20% in overstory trees in NEF, whereas in MAT-R, N concentrations in forbs and graminoids, the two dominant cover types, ranged from 1.54 to 2.28%. N content in legumes in MAT-R was somewhat higher, ranging from 2.12 to 3.55%, but legumes made up <5% of plant biomass (Ramsey 2002).

Table 6.

N budgets for TNEF, MAT-F, and MAT-R watersheds.

N inputs to the two watersheds from deposition were assumed to be equal based on their proximity. Mean annual nitrate export in stream water was much less from MAT than from NEF (0.11 vs. 4.64 kg $N \cdot ha^{-1} \cdot yr^{-1}$, respectively), suggesting that the reclaimed mine watershed is a much stronger sink for N than the forested watershed (Table 6). In fact, Castro et al. (2007) reported that in wet years, NEF watershed was a net source of N.

Within the soil, the net N mineralization rate was similar at NEF and MAT-R sites (Table 7). However, net nitrification rate and the percentage nitrification were significantly lower in MAT-R, meaning that nitrification was much less prevalent at the reclaimed mine site compared to NEF. Both ammonium and nitrate concentrations in the soil were significantly lower in MAT-R, which mirrors the pattern in total nitrogen concentration (see Table 2). Nitrification primarily depends upon a high concentration of ammonium in soil. Thus, the low ammonium concentration at MAT-R could explain the low nitrification rate.

Table 7.

Net N mineralization, net nitrification, and inorganic N concentrations (0–10 cm depth).

Phosphorus pools

Extractable P pools in soil were significantly lower at MAT-R than at the two forested sites in both horizons (Fig. 2; see also Simmons and Currie 2005). For example, in the mineral soil horizons, MAT-R contained 17.0 kg P/ha whereas NEF and MAT-F contained 66.4 and 82.3 kg P/ha, respectively. Similarly, total soil P was significantly lower at MAT-R than at the other two sites (Fig. 2), indicating that a substantial portion of the P pool was lost at MAT-R due to the land-use change.

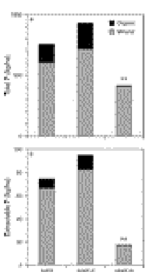

Fig. 2.

(a) Total soil P pools and (b) bicarbonate-extractable (plant-available) P pools in a forested watershed (NEF), the forested portion of a mined watershed (MAT-F), and the mined and reclaimed portion of a mined watershed (MAT-R). Pools were measured 15 years after reclamation was completed to a depth of 10 cm.

** $P < 0.01$.

Hydrology

Computed annual runoff values for the stations along TNEF1 and TMAT1 reveal significant variations in hydrology among the 2000, 2001, and 2002 water years (Fig. 3). It should be noted that this study period was exceptionally dry for western Maryland with below-average rainfalls in all three years. From TNEF1, annual runoff in years 2000, 2001, and 2002 ranged from 172 to 302 mm, whereas in TMAT1 annual runoff in the same years ranged from 217 to 271 mm. Compared to a long-term runoff value of 390 mm based on nearly a century of data from the Georges Creek at Franklin (Maryland) station, these values were between 23% and 56% below normal. As a percentage of incident precipitation annual runoff from the two watersheds was similar, ranging from 18 to 32% and 23 to 28% for NEF and MAT, respectively. Likewise, evapotranspiration percentages were the same (68–82% at NEF and 72–77% at MAT).

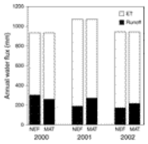

Fig. 3.

Annual water flux (evapotranspiration [ET] and runoff) from NEF and MAT watersheds for water years ending in 2000–2002. The height of each bar represents annual precipitation.

However, on a shorter time scale the two watersheds behaved very differently. Comparison of the runoff hydrographs (normalized by watershed area) for the two stream stations shows similar responses to precipitation in terms of timing, but very different responses in terms of magnitude (Fig. 4). For most events (even snowmelt), storm-flow peaks occur on the same dates. The major difference in response is that the magnitude of peak runoff in TMAT1 is consistently greater than in TNEF1. Negley and Eshleman (2006) observed that TMAT1 peaks were typically a factor of 2–10 times greater than the peaks at TNEF1. Similar differences in response can also be seen by visually comparing flow–duration curves, which indicate that TMAT1 tended to exceed TNEF1 during high flow and during very low flows but that at intermediate flow conditions the reverse was true (Fig. 4b). Thus, although the annual runoff of the two streams was similar, TMAT1 discharged more water than TNEF1 during storm events but less during base flow conditions.

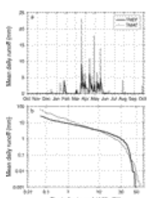

Fig. 4.

Mean daily runoff of TNEF1 and TMAT1 (a) vs. time and (b) vs. exceedence probability (log–log scale), both for the 2000–2001 water year.

Stream chemistry and temperature

Despite the mining that occurred in MAT watershed, few symptoms of acid mine drainage were apparent in the stream (although a few, small iron-rich seeps were identified at the site). Spot checks in 2002 showed slightly elevated conductivities ranging from 86 to 153 μS/cm in TMAT1 compared to 37–55 μS/cm in TNEF1. However, there was no difference in stream pH. Weekly samples of TMAT1 from 2000 to 2002 water years ranged from 4.04 to 7.33 with a mean of 6.39. TNEF1, in comparison, ranged from 3.59 to 7.15 with a mean of 6.37 (K. N. Eshleman, *unpublished data*).

Still, several notable differences in stream chemistry were apparent. Daily mean sediment concentration was 44.5 mg/L in TMAT1, significantly higher than the 15.7 mg/L in TNEF1 (paired t test, $P < 0.05$). It is likely that the higher storm flows in TMAT1 and smaller biomass of "protective" vegetation explains the higher sediment transport in this watershed. Related to the large sediment flux was a 1.5–2.3 times larger export per unit area of total P and particulate P from MAT watershed (Table 8). Particulate P is part of the sediment load and is the largest component of total P at this site. Similarly, DOC flux was substantially greater in TMAT1 in all three years (Table 8).

Table 8.

Annual stream fluxes from a forested watershed (NEF) and a reclaimed mine watershed (MAT).

In stark contrast, the nitrate export rates per unit area in TMAT1 were only 0.9–3.1% of those from TNEF1, meaning that MAT watershed was a strong sink for this ion (Fig. 5). This suggests that the watershed is N limited and that plants and microorganisms are rapidly immobilizing mobile forms of N.

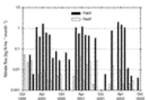

Fig. 5.

Monthly runoff flux of nitrate from TNEF1 and TMAT1 from October 1999 through September 2002. Note the *y*-axis log scale.

Stream temperatures of TNEF1 and TMAT1 followed a similar seasonal pattern, but the absolute values were distinctly different from each other. Mean daily stream temperatures ranged from 0.8° to 22.8°C at TNEF1 and from −3.8° to 39.8°C at TMAT1. Mean monthly stream temperatures in TMAT1 were on average 3.5°C higher from May through August and 7.0°C lower from November through February (Fig. 6a). As with soil temperatures, the amplitude of daily fluctuations in stream temperature was greater in TMAT1 than in TNEF1 from March through October, especially during the summer months (Fig. 6b). Apparently the canopy cover provided by the trees in NEF moderates temperature swings in the stream as well as in the soil, keeping them from being as extreme as they are in MAT watershed.

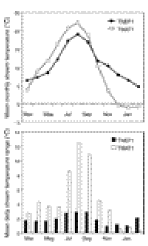

Fig. 6.

Mean monthly stream temperature and mean daily range of stream temperature for TNEF1 and TMAT1 from March 2001 to February 2002.

Aquatic leaf decomposition

Leaf litter decomposition is an indicator of the activity of the macrofaunal, meiofaunal, and microbial communities in a stream. TNEF1 exhibited the highest decomposition rates during all three incubations ranging from 0.0123 to 0.0364 d^{-1} (Fig. 7). Decomposition in TMAT1 was much lower, 0.0013–0.0079 d^{-1}. The two sites in Matthews Run, which receive some acid mine drainage from other sources, usually had rates that were intermediate.

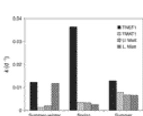

Fig. 7.

Mean leaf decomposition rates (*k*) from a forested watershed (TNEF1), from a reclaimed mine (TMAT1), from the confluence of TMAT1 and Matthews Run (L. Matt), and from Matthews Run above the confluence (U. Matt). The summer–winter incubation period was July 2001 through March 2002 (228 days), the spring period was March through July 2002 (129 days), and the summer period was July through October 2002 (96 days).

Macroinvertebrate communities in the harvested leaf packs also varied among sites. Compared to TNEF1, the sites in Matthews Run contained a smaller percentage of sensitive species and a slightly larger percentage of tolerant species (Table 9). The leaf packs in TMAT1 housed substantially fewer organisms and a larger percentage of tolerant species than those from the other sites. Apparently, the mining and reclamation process has had an effect on macroinvertebrate populations in TMAT1. The slower decomposition rates in TMAT1 could be explained by daily spikes in stream temperature (some as high as 39°C) and by long periods when the stream channel is dry, both of which negatively affect stream biota. Williams et al. (1995) similarly reported reductions in macroinvertebrate and fish diversity due to mining operations in a mined stream in western Pennsylvania; although in that case, acid mine drainage was likely the main cause for the reduction in species.

Table 9.

Mean macrofaunal metrics at four locations on four sampling dates between 1999 and 2002.

Discussion

Based on soil maps, aerial photos, permit records, and their close proximity, it is highly probable that these two ecosystems, NEF and MAT, shared a similar ecosystem structure and function prior to 1981. Then the two watersheds were subjected to two different management regimes. The soil disturbance in MAT-R was much more severe than in NEF because of the removal, storage, and reapplication procedures during which soil material became mixed and homogenized. Previous studies of mine soils have demonstrated the reduction in soil organic matter and nitrogen (Davies et al. 1995, Boerner et al. 1998, Akala and Lal 2000, 2001) and loss of soil biotic diversity and activity (Scullion et al. 1988, Harris et al. 1989) during topsoil storage. As a result the biogeochemistry of MAT-R was radically altered. In contrast, in MAT-F where only clear-cutting occurred, the small trees, seedbank, and soil horizons were left intact. Consequently, twenty years later we find that MAT-F is more similar to NEF than to MAT-R in all aspects compared in this study. The difference in recovery between MAT-R and MAT-F is fundamentally due to a difference in disturbance severity; MAT-F is undergoing secondary succession, whereas MAT-R is undergoing primary succession. A key implication of this is that recovery of MAT-R will take much longer.

Our results show that the carbon and nutrient cycles of the reclaimed mine site are dramatically different. Carbon storage was substantially reduced at MAT-R mainly because of the removal of woody vegetation. Remarkably, underground C storage at MAT-R was very similar to that of the forested sites despite the removal and replacement of soil during the mining operation. Almost universally, mining and reclamation by conventional methods results in a soil low in C and N, and we have no reason to believe that MAT-R was any different (Roberts et al. 1988, Simmons and Currie 2005). Nevertheless, the vegetation at MAT-R was able to replenish its mineral horizon and total belowground carbon pools in the intervening 15-year period to the point where they equal 96% and 73%, respectively, of NEF's pools. This rapid replenishment has been documented at other mining sites with rates ranging from 300 to 1300 kg C·ha^{-1}·yr^{-1} and is likely due to litter inputs and underground productivity of fine roots (Roberts et al. 1981, 1988, Akala and Lal 2000, 2001). At these rates, most of the soil carbon in a typical temperate ecosystem could be replenished in 20–50 years. Similarly, McLauchlan and colleagues (2006) reported a C accumulation rate of 620 kg·ha^{-1}·yr^{-1} for grasslands converted from agricultural fields in the Great Plains and a C pool recovery time of 55–75 years. Knops and Tilman (2000) argued that C accumulation in old-field succession was controlled by N accumulation, that is, that C accumulation was linked to productivity which was controlled by soil N. Thus, there are likely to be feedbacks between C accumulation and N accumulation.

Nevertheless, managers should be circumspect in determining when a soil's C resources are recovered. For example, replenishment of deep soil C reserves may be slower; Schafer et al. (1980), studying western soils, reported that below 20 cm, carbon replenishment could take as long as 400 years. Another caveat is that although total soil C may be restored within a few decades, soil C distribution and dynamics may still differ from native soils. Specifically, these “young” soils have less aggregate structure, low microbial biomass, and less carbohydrate carbon (Malik and Scullion 1998).

Perhaps the greatest distinction between the reclaimed mine site and the forested sites lies in the nitrogen cycle. Castro et al. (2007) evaluated the N dynamics of NEF over a five-year period and concluded that it exhibited several symptoms of N saturation: high concentrations of nitrate in stream water, high concentrations of N in foliage, wood, and fine roots, low C:N ratios in the soil, high soil nitrification rates, and elevated soil solution N concentrations. Furthermore, during wet years, NEF acted as a net source of N to stream water. Although fewer measurements were made at MAT-F, it shares many of the characteristics of NEF, namely, elevated foliar, wood, and fine-root N concentrations, and low C:N ratios in the soil.

A comparison of the N budget of NEF with a well-studied forest system just 90 km away (the Fernow Experimental Forest) shows that NEF is not atypical. Watershed 4 at Fernow is a reference watershed that has been monitored since 1956. Mean annual N deposition and export from 1984 to 1993 were 11.3 and 5.22 kg N/ha, respectively (Adams et al. 1997). The N deposition rate at Fernow was higher probably because precipitation volume during that period was nearly 50% greater than at NEF (1451 vs. 983 mm). The N export from Watershed 4 is comparable to that of NEF.

In contrast to NEF, the mined portion of MAT exhibited lower N concentrations and higher C:N ratios in foliage (data not shown), very low nitrification rates, and much lower concentrations of N in stream water, suggesting that MAT-R is a N-limited ecosystem that is tightly conserving N. In NEF most of the ammonium produced by mineralization was quickly converted into nitrate, much of which then left the system via stream export. The low concentrations of extractable ammonium in MAT-R suggest that plants and microorganism are competing fiercely for the soil N resources. Nitrifying bacteria activity in soil is dependent upon a large, constant ammonium supply, which does not exist in MAT-R; thus, nitrification was minimal. With low concentrations of inorganic N and most of it in the relatively nonmobile NH_4^+ form, little was available for leaching into stream water.

In our study, net N mineralization did not differ significantly between NEF and MAT-R, but other researchers have reported much lower rates in reclaimed minesoils due to lower N availability

(Lindemann et al. 1989, Williamson and Johnson 1994, Davies et al. 1995). A key difference is that we report our mineralization rates on a unit area basis with the result that the high bulk density of MAT-R soils tends to compensate for lower mineralization rates expressed on a mass basis.

So it appears that the mining and reclamation process altered the N cycle of MAT-R, changing it from a net source or weak sink of N to a strong N sink. Removal of a large amount of nutrient capital in the form of woody biomass and probable losses of N during reclamation (Davies et al. 1995) resulted in much smaller ecosystem N pools immediately after reclamation was completed. Currently, there is still less N per unit area contained in MAT-R than in NEF or MAT-F (Table 6). Rates of net N accumulation are controlled by many factors including presence of legumes and chemistry of spoil material used as well as atmospheric deposition rates (Vimmerstedt et al. 1989, Knops and Tilman 2000). We did not measure N fixation but legumes were present in the grassland and undoubtedly contributed some amount of N to the system. Another potentially important flux is loss of N through denitrification in the soils of the ephemerally inundated depressions at MAT-R. We did not measure denitrification in situ so it is not possible to estimate the amount of N lost from the ecosystem through this pathway. The mass balance for the ecosystem shows that MAT is accumulating a mean of 6.6 kg $N·ha^{-1}·yr^{-1}$ on a net basis from deposition inputs (Table 6).

During 15 years of recovery some of the N pools have been partially replenished. For example, MAT-R mineral soil N is equal to 79% of NEF and total belowground N is equal to 65% of NEF. However, other pools, like woody biomass and Oe/Oa pools, are either nonexistent or remain very small. Other studies have shown the rate of net N accumulation in reclaimed minesoils ranges from 21 to 79 kg $N·ha^{-1}·yr^{-1}$ (Shafer et al. 1980, Roberts et al. 1988). Knops and Tilman (2000) reported a rate of 12.3 $kg·ha^{-1}·yr^{-1}$ net N accumulation in abandoned agricultural fields.

The soil phosphorus pools also showed striking differences between the reclaimed mine site and the forested sites. Simmons and Currie (2005) reported that there was significantly less total P in MAT-R soils than in NEF soils (1230 vs. 1810 kg P/ha, respectively). Soil organic P and plant-available P pools were also significantly lower. They further concluded that replenishment of total P from weathering proceeded much more slowly than replenishment of C or N. A comparison of mineral soil replenishment after 15 years illustrates this disparity. Compared to the native soil pools, MAT-R mineral soil contained 95% of the C but only 79% of the N and 69% of the P. This suggests that during the early stages of recovery the soil C:N:P ratio will tend to widen until C pools are restored. After this point, the C pool will remain relatively constant while N and P pools continue to accumulate leading to a gradual narrowing of the ratio toward the native soil ratio. In the case of P, with typical accumulation rates of 1–2 $kg·ha^{-1}·yr^{-1}$, complete recovery may take centuries (Newman 1995, Simmons and Currie 2005). Other studies have shown that plant-available P can decrease with time after reclamation, as freshly exposed Fe and Al minerals in these young horizons rapidly weather, leading to increased P adsorption capacity (Roberts et al. 1988). All of this suggests that P may be a limiting factor to plants and microorganisms for the first several decades of recovery.

Admittedly, the design of this experiment could be considered pseudoreplication; however, this has been accepted as the nature of long-term, paired watershed studies since the initial Hubbard Brook Ecosystem experiments (Likens et al. 1977). The recognized trade-off in these studies is an intensive, cumulative, long-term data set at the expense of large sample size. This approach has been valuable in a wide range of studies including studies of soil C storage (Ross et al. 1999), hydrology (Williams et al. 1995), N dynamics (Peterjohn et al. 1996, Foster et al. 1997, Magill et al. 1997, Jefts et al. 2004), acid deposition (Driscoll et al. 1996, Adams et al. 1997), and logging (Keppeler and Ziemer 1990).

Mining and reclamation appeared to have had profound impacts on the hydrology of the ecosystem as well. Grading and the use of heavy equipment during the reclamation process compacted the surface soil and lowered the infiltration capacity of the soil and resulted in more storm runoff and, therefore, a “flashier” stream (Negley and Eshleman 2006). In between rain events this ephemeral stream rarely carried water.

However, the annual runoff and percentage annual runoff was very similar to the forested watershed, NEF, despite the fact that MAT-R had no canopy to intercept rainfall and a much slower infiltration rate. A seven year study of Stony Fork watershed in western Pennsylvania showed similar results; no difference in total annual discharge between a control watershed and a watershed with 11.3% of its surface area disturbed by mining and reclamation (Williams et al. 1995). Why did only about one-quarter of the rainfall at MAT-R runoff into the streams? The answer probably lies in the microtopography of MAT-R which consists of small undulations formed by the grading process and subsequent settling of the fill material (Ramsey et al. 2001). The numerous depressions, mostly in the range of 1–10 m in size, served as catchment basins for rainfall and most likely prevented uniform sheet flow from carrying runoff all the way to the stream channel; instead, water was held by the depressions, creating ephemeral wetlands in some cases, until it either evaporated or slowly infiltrated into the soil. The consistently drier soil in MAT-R suggests that most evaporated. In contrast, the soil surface in NEF watershed was shaped by millions of years of erosion resulting in an interconnected network of rills that effectively funneled all of the runoff into the stream channel. Thus, the irregular microtopography of MAT-R mimics the water-holding capacity of the forest canopy by holding a fraction of the incident precipitation until it can be evaporated.

The tall, narrow storm peaks evident in TMAT1 hydrographs were likely caused in part by the minimally vegetated riparian zone. Lack of a tree-dominated buffer zone would also explain the higher and more variable stream temperatures in TMAT1 as well as the higher sediment concentrations. High stream temperatures coupled with tall, narrow flood peaks and elevated sediment concentrations may have contributed to the reduced number of sensitive macroinvertebrates in the streambed and the reduced leaf decomposition rates.

Thus, changes to the terrestrial ecosystem have had major impacts on the adjoining stream ecosystem in two ways. First, changes to the physical environment (i.e., reduction in leaf area, undulating microtopography, and soil compaction) have altered the stream's hydrological characteristics and increased the sediment concentration. Second, changes to the biogeochemical cycles of N and P have drastically reduced the amount of N but increased the amount of P entering the stream. These changes have important implications for downstream eutrophication, particularly in the Chesapeake Bay watershed.

Land-use change inevitably leads to changes in ecosystem structure and function; however, the direction and degree of change depends on the type of conversion. There are some parallels between a forest to mineland conversion and a forest to urban land-use conversion. In both cases soil permeability decreases; in one case due to soil compaction and in the other due to addition of buildings and pavement. Streams in urban areas share a set of unique characteristics that have been called the urban stream syndrome. Some of the reported symptoms include a flashier hydrograph, elevated concentrations of nutrients, altered channel morphology, reduced biotic richness, increased dominance of tolerant species, increased stream temperature, and a decrease in leaf breakdown (Walsh et al. 2005, Chadwick et al. 2006). In our study, TMAT1 exhibited all of these symptoms with the exception of elevated stream N concentrations.

These similarities in stream condition most likely derive from the reduction of forest biomass in both types of watersheds and decreased soil permeability. Reduction in forest biomass, especially in riparian zones, means less water uptake and less shade which could contribute to the observed flashier streams and warmer water temperatures. Decreased permeability leads to flashier streams and greater erosion and transport of particulate matter.

Effects on stream nutrient concentrations are less straightforward. In MAT stream water, N concentrations were low because of a reduction in soil N concentrations. Soil P concentrations were also low, but higher rates of erosion of particulate P during storm events led to an overall increase in stream water P transport. In contrast, in urban areas N and P inputs from soil are often overshadowed by much larger inputs from storm water systems, sewage leaks, and industrial discharges (Walsh et al. 2006).

Because we studied only a single mined watershed, we cannot with any surety extrapolate to a regional scale. In order to be able to predict with confidence the consequence of this increasingly common land use, conversion studies of more mined watersheds are needed. These should include mines from a variety of locations using a variety of mining and reclamation techniques and the studies should examine both short- and long-term changes.

We have documented both local (within-watershed) and downstream impacts 15 years after a forest to reclaimed mine conversion. Effects on biogeochemical cycles were profound and in some cases long-lasting. Soil C, N, and P pools were recovering at nonuniform rates that could lead to nutrient limitations over decades and centuries. Hydrologic impacts were also apparent in the form of taller, narrower storm peaks which caused increased risks of flooding and increased loads of sediment and particulate P. In the mined stream, N concentrations and litter decomposition were reduced, but P concentrations and temperature were higher. These fundamental changes in the stream environment will affect stream nutrient cycling and carbon processing with impacts on aquatic flora and fauna downstream.

Currently the goal of mine reclamation is simply the establishment of permanent vegetative cover. This approach is shortsighted and does not take into account the importance of ecosystem processes like nutrient cycling nor the potentially harmful conditions created, like high soil and stream temperatures. As a result, recovery of comparable ecosystem function will take decades to centuries. Adverse impacts of surface mining could be minimized by making restoration (reestablishment of a similar ecosystem), rather than reclamation, the goal. Restoration requires a longer term view with a focus on reestablishing ecosystem processes and most components of the community. Practices such as adding soil organic matter, planting native species, and avoiding soil compaction would hasten the recovery of the ecosystem and the reestablishment of basic ecosystem functions.

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