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COMPENSATING FOR WETLAND LOSSES UNDER THE CLEAN WATER ACT

Committee on Mitigating Wetland Losses

Board on Environmental Studies and Toxicology

Water Science and Technology Board

Division on Earth and Life Studies

National Research Council

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Preface

The U.S. Army Corps of Engineers (Corps) and the U.S. Environmental Protection Agency (EPA) share responsibility for regulating the mitigation (lessening of impacts) of damages to wetlands. In response to a request from EPA, the National Research Council (NRC) formed the Committee on Mitigating Wetland Losses to evaluate mitigation practice as a way to restore and maintain the quality of the nation's waters, particularly as regulated under Section 404 of the Clean Water Act.

The committee reviewed the available literature on replacement of wetland functions, considered both restoration and creation efforts, visited several mitigation sites around the United States, and then evaluated both the ecological performance of mitigation projects and the institutions under which mitigation projects are conducted (permittee-responsible mitigation banks and in-lieu fee programs). At a series of five meetings, the committee worked in a truly interdisciplinary and collaborative manner to develop the conclusions and recommendations presented in this report.

The committee is grateful for the briefings and the assistance provided by the staff of EPA, the Corps, the U.S. Fish and Wildlife Service, and the National Marine Fisheries Service.

The committee is also grateful for the excellent and untiring support provided by the NRC staff, who organized the meetings and field trips and kept us on track in addressing the major tasks, as well as the fine details in report preparation. Dr. Suzanne van Drunick, our outstanding project director, kept the process on track and made sure that the report

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was coherent. We all benefited greatly from the help of Jennifer Saunders, who followed Leah Probst as project assistant. Ruth Crossgrove, Milenada Karalic-Loncarevic, and Barbara O'Hare helped with the many details that made the report ready for publication. Dr. David Policansky initiated the project, and we thank him for providing stimulating discussions. Dr. James Reisa's suggestions improved the Executive Summary.

The committee members were exemplary in their dedication to this complicated task; without their expertise, hard work, and timely responses, completion of the project would not have been possible.

Joy B. Zedler
Chair, Committee on Mitigating Wetland Losses

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2

Outcomes of Wetland Restoration and Creation

INTRODUCTION

Underlying wetland mitigation is the assumption that it is scientifically possible for humans to recreate the structure and functions of a wetland, either by restoring a site that had previously been a wetland or by creating an entirely new wetland. The purpose of this chapter is to discuss the ecological principles of wetland creation and restoration science and evaluate the current scientific ability of practitioners to restore or create various aspects of wetland functioning in a variety of environments. The chapter is structured to answer several questions posed in the committee's Statement of Task about the ecological basis of wetland mitigation.

Is It Possible to Restore or Create Wetland Structure?

Wetland Types That Have Been Restored and Created

The committee evaluated restored and created wetlands from around the United States, including coastal and inland projects, by reviewing numerous scientific studies (see Appendix A) and by visiting several wetland mitigation sites. The following findings are based on the committee's analysis.

Many types of herbaceous wetlands have been restored or created to a condition that appears to replicate natural wetland structure, such as freshwater emergent marshes (Lindau and Hossner 1981; Niswander and

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Mitsch 1995; Wilson and Mitsch 1996; Brown and Bedford 1997) and wet meadows (Brown and Veneman 1998). Mixed results have been reported in the restoration of wet prairies and sedge meadows (Galatowitsch and van der Valk 1996; Ashworth 1997). Certain floristic assemblages, such as sedge meadows visited by the committee in the Chicago area, require extensive planting and intensive management in order to maintain the desired species composition. Thus, the technical ability to attain a prescribed species assemblage may not be affordable or practical in the long term.

Shrub swamps and forested wetlands are more difficult to create or restore because of the time needed to establish mature woody plants (Niswander and Mitsch 1995; Brown and Veneman 1998; King 2000). The committee observed examples of created wetlands where tree saplings had been planted and appeared to be viable, but forest structural characteristics (e.g., stand density, stand height, basal area per tree) were quite different from those of the mature stands they were intended to replace. Planted trees are usually small in diameter, so that basal area per tree is small in comparison to natural forested wetlands. The density (trees per unit area) of planted stands is typically higher than that of natural stands because of either permit specifications or the desire to compensate for mortality. Given sufficient time, planted trees would be expected to attain basal areas comparable to those of trees in natural stands, but densely planted stands would continue to differ from natural stands unless thinned.

Seagrasses and salt marshes are sometimes described as wetlands that are relatively easy to restore or create, based on a long history of mitigation involving eelgrass (*Zostera marina*) and smooth cordgrass (*Spartina alterniflora*; Simenstad and Thom 1992; 1996). Each of these natural vegetation types has the following features that are amenable to restoration or creation:

- One vascular plant species dominates the vegetation; that is, it forms the matrix of the ecosystem.
- The species in question have been well studied for growth requirements and environmental tolerances.
- The matrix species is readily collected and propagated and planted because of its particular clonal growth form (ramets are produced from rhizomes that are easily subdivided).
- These vegetation types grow in relatively wet conditions where environmental variability is buffered by ocean water of relatively constant temperature, salinity, and pH.
- These species are natural colonizers of bare substrates ("early succession" species).

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- Transplantation can be done with considerable assurance that environmental conditions will not change substantially after planting (relatively low risk of extreme events, high probability of establishment).
- Microbial and animal components of these ecosystems are readily dispersed by the widely circulating ocean waters that connect distant seagrass beds and/or salt marshes.

If these are indeed the characteristics that make ecosystems easier to restore or create, one can expect greater difficulty in restoring ecosystems that have the following:

- Several codominants.
- Poorly studied species.
- Species that are dispersal limited or that have low reproductive capacity (few seeds, heavy seeds, no vegetative reproduction).
- Dominants that are "late succession" species (not ready colonizers of bare substrates).
- Habitats that have high environmental variability, which allows exotics an opportunity to invade and which pose risks for transplantation efforts (e.g., a marsh plain that is exposed for much of the day can become dry and hypersaline the day after planting, stressing and killing thousands of seedlings (J. Zedler, University of Wisconsin, personal observation, 2000)).
- No aquatic connection to other wetlands and/or no corridors for dispersal.

The restorability of an ecosystem is not necessarily predictable from experiences with smooth and eelgrass cordgrass, even for closely related habitat types. For example, restoration of tall forms of Pacific cordgrass has proved to be difficult because *Spartina foliosa* has a high nitrogen demand and grows poorly in substrates that are coarse in texture and/or low in organic matter (Langis et al. 1991; Zedler 1998). Likewise, seagrass beds in Florida that are dominated by late-succession *Thalassia testudinum* are difficult to restore because this species grows slowly and not densely enough to prevent erosion (Fonseca et al. 1998). Moving upslope to mid- and high-intertidal wetlands complicates restoration and creation efforts even more, as the environment becomes highly variable in soil moisture and soil salinity. In addition, newly planted vegetation becomes susceptible to a broader range of herbivores in that both aquatic and terrestrial animals can attack delicate ramets (Zedler 2001).

Wetland Types That Are Difficult to Restore or Create

Wetland ecosystems that require a specific combination of plant types, soil characteristics, and water supply are difficult to impossible to create from scratch. Examples include vernal pools, fens, and bogs.

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Vernal pools

The term "vernal pool" is widely applied to small shallow depressions that hold water for brief periods of time. The vegetation and invertebrates are specific to the regions where such temporary pools form, and the timing of pool formation is also variable. In the forests of the northeastern states, such pools form in spring and fall. In the arid southwestern states, they form during the winter rainy season.

Attempts to restore and create vernal pools in Southern California indicate the importance of locating such efforts where the substrate can support temporary ponding. Restoring or creating vernal pools begins by re-creating the topography where there is appropriate substrate (a clay layer) that seals upon wetting. There, the hydroperiod is critical to restoration or creation of native vegetation and fauna; if the hydroperiod is too long, cattails will invade and outcompete the more ephemeral vascular plants (P. Zedler, San Diego State University, personal commun., 2000). Predators, such as fish, might invade and eliminate the macroinvertebrates (such as fairy shrimp). In a long-term study of California vernal pools that were created by excavating depressions near natural pools, the hydroperiods did not converge with those of the reference systems until year 10 (Zedler al. 1993). Seeds, spores, and resting stages of the plants and animals can be vacuumed from natural pools and used to inoculate restored and natural pools, producing new vegetation highly similar to that of the natural site if the hydroperiod is correct. Even the endangered mesa mint (*Pogogyne abramsii*) has been reestablished at created pools of Southern California. However, few data are available on the similarity of algal and animal components of vernal pools.

Perhaps the greatest difficulty is that vernal pool landscapes cannot be replaced in areas like California, where vernal pools occur on relatively flat topography, which is prime real estate. When the remaining vernal pools are destroyed, there is no place to re-create them except by increasing the density of pools in other remnant landscapes. It is unclear how overall system function is altered by increasing the density of pools while decreasing the landscape area that vernal pools occupy. Another problem is that the uplands surrounding vernal pools also are in short supply and support rare species. Thus, as the available land decreases, vernal pool creation and restoration increasingly conflict with efforts to preserve habitat for other endangered plants and animals.

Fens

Fens are herbaceous wetlands that develop on calcium-rich organic soils and for which groundwater seepage is an important water source. Fens are among the most species-rich wetlands in North America. Their

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plant diversity is high per unit area, yielding long species lists; approximately 100 vascular plant species might occur in a wetland, with 15 to 30 or more species per square meter (Bedford et al. 1999). Typically, a variety of species share dominance, rather than a preponderance of any one species. Oligotrophic conditions occur because fens are fed by groundwater that is low in nutrients (nitrogen and phosphorus) and high in calcium. Calcium is important, not only for its effect on pH but also because it precipitates out phosphorus, lowers nutrient availability, and thereby reduces the chances that any one species will outcompete its associates.

Drainage of fens changes both the hydrology and the soil chemistry. Exposure to surface-water runoff also changes soil chemistry. In both cases, nutrients are released to the fen, allowing opportunistic species the competitive advantage. Draining also exposes any sulfides to oxygen, forming sulfuric acid, which lowers pH and prevents the restoration of calciphilic vegetation. van Duren et al. (1998) reported slightly decreased pH for drained fens in the Netherlands, where additions of lime (calcium carbonate) were ineffective in reestablishing calciphiles. Vegetation was judged nonrestorable, even when surface-water runoff was entrained in a "treatment wetland" just upstream of fen. Likely constraints on restorability were either inflowing nutrients that escaped treatment or persistent acidic soil.

Bogs

Bogs occur on acidic organic soil ("peat") that develops over millennia from the accumulation of plant decomposition remains. In eight studies summarized by Johnston (1991), natural peat accretion rates ranged from 0.1 to 3.8 millimeters (mm) per year, which indicates an extremely slow rate of development. Bog drainage exposes the organic soil to aeration, accelerating decomposition and fundamentally altering organic carbon compounds in the soil. Agricultural uses of bogs further alter soil chemistry and structure through tillage, fertilizer inputs, and subsidence as soils compact and oxidize. Peat fires can oxidize in days the organic carbon that has taken centuries to accumulate. Although vegetative cover has been reestablished on bogs that have been subjected to such extreme losses and alterations of substrate, restoration of original plant communities is extremely difficult (Mitsch and Gosselink 2000).

The harvesting of live sphagnum moss from bog surfaces is a small but viable industry (Johnston 1988). Research has demonstrated that natural recovery of the moss surface following harvesting takes about 20 years (Elling and Knighton 1984). In contrast, reclamation of wetlands mined for peat has been very difficult because (1) surface mining causes major changes in local hydrology, (2) peat accumulates at a very slow rate, and

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(3) the chemistry of old peat is quite different from that of surficial peat layers that support bog vegetation (Updegraff et al. 1995).

The committee concludes that some types of wetlands can be restored and/or created (e.g., freshwater emergent marshes) but that others cannot (e.g., fens and bogs). Not all emergent marshes will be easy to replace. Some types (e.g., species-rich sedge meadows), some hydrological contexts (e.g., late-summer drawdowns), and some functions (e.g., biodiversity support, especially species with narrow ranges of ecological tolerance) will likely be more challenging to reproduce than others (e.g., cattail marshes, continuous flooding, and high plant productivity, respectively). Some types of wetlands will be more difficult to restore or create in certain settings (i.e., where landscape positions, specific substrates, or adjacent land uses are inappropriate).

Are Wetland Functions Replaceable?

Wetlands provide a number of ecological functions. The three most commonly cited wetland functions are related to water quality, hydrology, and habitat, but other functions also exist (e.g., alteration of microclimate, carbon sequestration). Some ecological functions provide human benefits, such as improvement of downstream water quality, whereas others may benefit only nonhuman organisms (i.e., wetland flora and fauna). Knowledge of the existence of wetland functions increases with increasing scientific understanding, but the perceived importance of different wetland functions changes as human values change. For example, the carbon-sequestration function of wetlands has recently assumed increased importance with our increased understanding of the role of atmospheric trace gases in global climate change (Bridgham et al. 1995).

The establishment of wetland structure does not necessarily restore all the *functions* of a wetland ecosystem. For example, denitrification (an ecological process that benefits water quality) requires the presence of nitrate supply, a labile carbon source, anaerobic conditions, and microbial activity. Thus, a site that has wetland structure in terms of its vegetation assemblage might not provide the function of denitrification if these four requirements are not met.

FIVE WETLAND FUNCTIONS

The following sections discuss five major wetland functions that warrant attention in evaluating wetland restoration and creation: hydrological functions, water-quality functions, support of vegetation, support of habitat for fauna, and soil functions. These do not represent all wetland functions but do include a number of important ones that are frequently

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overlooked in wetland functional replacement. Functional assessment of wetlands proposed for evaluating the development of mitigation sites is discussed in Chapter 7.

Hydrological Function

Hydrology is most often cited as the primary driving force influencing wetland development, structure, function, and persistence. Consequently, establishment of the appropriate hydrology is fundamental to wetland mitigation through either restoration or creation. Hydrological processes influence water quality through nutrient inflow and outflow from the system (Gosselink and Turner 1978; LaBaugh 1986; Carter 1986; Day et al. 1988; Novitzki 1989) and by creating an environment (soil saturation) that allows anaerobic conditions to develop and reducing chemical reactions to operate. Reducing conditions in the soil causes denitrification to occur, organic matter to accumulate, and chemical transformations of phosphorus and iron that influence their solubility (Woodwell 1956; Gosselink and Turner 1978; Richardson et al. 1978; Riekerk et al. 1979; Sharitz and Gibbons 1982; Carter 1986; LaBaugh 1986; Wilcox 1988). Hydrology also influences seed distribution (Gosselink et al. 1990; Sharitz et al. 1990), seed germination, and species establishment and composition (Christensen et al. 1981; Carter 1986; Day et al. 1988; Gunderson et al. 1988; Conner et al. 1990; Gosselink et al. 1990; Sharitz et al. 1990). For many species, seedling recruitment and establishment are a complex process that is only partially understood. Of the seed available at a microsite, germination and establishment of most species are flood/inundation sensitive (Christensen et al. 1981; Day et al. 1988; Duever 1988; Gunderson et al. 1988). Very few species germinate in standing water. Inundation cessation is climatically controlled except where structural devices are used to manage hydrology. Once established, most species tolerate a fairly wide range of hydrological and other environmental conditions (e.g., light, nutrient availability, and soil-water chemistry).

In palustrine nonriverine systems, the hydrological variability that results in species diversity is caused by minor variability in the microtopography (Harper et al. 1965; Christensen et al. 1981; Daniel 1981; Hardin and Wistendahl 1983; USFWS 1983; McDonald et al. 1983; Duever 1988; Gunderson et al. 1988; Titus 1990). Vegetation, in turn, influences hydrology by retarding flow (Gosselink and Turner 1978; Gosselink et al. 1990; Sharitz et al. 1990) and influencing evapotranspiration and affecting soil and water chemistry by leaf litter, transport of oxygen, and other biological processes (Day 1982, 1983; Day et al. 1988).

The difficulty of restoring wetland hydrology increases as the degree of wetland degradation increases (Long et al. 1992). Many wetland resto-

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ration efforts have not established the appropriate hydrology (Kusler and Kentula 1990; Pfeifer and Kaiser 1995; Galatowitsch and van der Valk 1996).

One measure of effective restoration or creation is establishment of jurisdictional hydrology. Taking a conservative stance, the U.S. Army Corps of Engineers (Corps) 1987 Wetland Delineation Manual established the 5% criterion (see Box 2-1) as the jurisdictional threshold, a quantitative value that was reaffirmed by the NRC (1995). Since wetland hydrology is fundamental to wetland structure and function, those who implement restoration and mitigation projects have also tended to take a conservative stance, that is, to err toward the wet end of the transition zone (12.5% inundation or saturation during the growing season; see Box 2-1). The structure and character of a wetland that is inundated or saturated 5% of the growing season differ greatly from those of one that is inundated or saturated 12.5% of the growing season. The consequence of this mitigation approach has been the establishment of wetlands that are much wetter than normal for the given landscape position (Cole and Brooks 2000a) or a shift from intermittently inundated or saturated to having open water (Kentula et al. 1992a).

Water-Quality Improvement

Water-quality functions can be mitigated but rarely duplicated. For duplication to happen, the mitigation wetland would have to be of exactly the same wetland type with the same hydrological inputs and the

BOX 2-1
Duration and Timing of Inundation or Saturation

Clark and Benforado (1981) divided the hydrological continuum into six functional categories: permanently inundated (inundation >2 meters (m) 100% of the time), semipermanent (inundation <2 m 75-100%), regularly inundated or saturated (25-75%), seasonally inundated or saturated (12.5-25%), irregularly inundated or saturated (5-12.5%) and intermittently or never inundated or saturated (<5%). They suggested that areas saturated less than 5% of the growing season clearly exhibited upland hydrological characteristics and that areas saturated more than 12.5% clearly exhibited wetland hydrological characteristics. Inundation between 5% and 12.5% of the growing season represented the transition zone, with some landscapes in this category exhibiting upland characteristics and others being more characteristic of wetlands.

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same chemical (including sediment) composition. It is entirely possible for the restoration or creation site to have water-quality functions superior to those of the impact site. If the impacted wetland is a mineral or organic soil flat (Brinson's 1995 classification), it would make only a passive contribution to water quality (Evans et al. 1993), because its only water input is rain, and the water-quality function is simply to provide an area of runoff where both the surface and the subsurface drainage waters are relatively uncontaminated with pollutants. If a mitigation site is a restored riparian wetland located between a stream and a nonpoint pollution source (either urban or agricultural), the mitigation wetland will have a water-quality function superior to the impact site. However, if the impact site is a riparian wetland while the mitigation site is on a flat, the vast majority of the water-quality function of the impact site is lost. To determine the water-quality function of either the filled area or the mitigation site, it is necessary to make some assessment of both the quality and the quantity of groundwater and surface water entering the wetland (Hill 1996; Hill and Devito 1997; Bedford 1999). Different regions of the United States would need to evaluate both the quantity and the quality of water entering a wetland in order to assess the potential for water-quality improvement.

Support of Vegetation

Wetlands fail to support plant biodiversity when the environment is extremely hostile (e.g., extremely contaminated or hypersaline) or when one or a few species dominate the site. Monotypic vegetation can be formed by native species or exotic species. Cattails (*Typha* species and hybrids) are notorious for overtaking nutrient-rich wetlands (Wilcox et al. 1984), as are giant reed grass (*Phragmites australis/communis*), purple loosestrife (*Lythrum salicaria*), and reed canary grass (*Phalaris arundinacea*).

Invasiveness is a function of both the invader and the habitat it colonizes. Plants that invade wetlands are typically species with high seed production, high germination rates, and the ability to spread vegetatively. Seedling establishment is often the limiting factor, but once established, the clone can expand, such that a clone from a single seedling could come to dominate an entire site. An additional attribute of invasive species is their ability to take up and utilize nutrients from high concentrations in the water or soil supply. In the Everglades, for example, native *Cladium jamaicence* is adapted to oligotrophic waters; it absorbs and stores nutrients in leaf bases. In contrast, the invasive *Typha domingensis* takes up nutrients and distributes them throughout the plant, growing to greater heights and biomass, thereby outcompeting *Cladium* where surface waters are eutrophic (Miao and Sklar 1998).

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The habitat being colonized will be more invasible if there are microsites available for seedling colonization. Small gaps in the canopy or minor soil disturbance might be all that is needed to allow seeds to establish (Hobbs and Huenneke 1992; Lindig-Cisneros and Zedler 2001). Thus, human- or animal-caused disturbances can lead to establishment events. The combination of canopy gaps and nutrient inflows can virtually guarantee the establishment and spread of invasive species. Wetland restoration and creation sites are very susceptible to species invasions when (1) they are devoid of vegetation, (2) plant canopies have multiple gaps, and (3) their water supplies are eutrophic. All three attributes characterize many mitigation sites.

The ability of a site to support biodiversity is not independent of its ability to improve water quality. It is critical that the relationship between these two functions be understood if mitigation goals are to be set that are ecologically conflicting, such as maintaining high plant biodiversity *and* improving eutrophic water quality. Mitigation sites that receive nutrient-rich surface-water runoff are well situated to perform water-quality-improvement functions, but biodiversity-support functions may suffer in the process. Wetlands that are designed to maximize the water-treatment function typically become monotypes of invasive species within a few years, even if they are initially planted to multiple species (Kadlec and Knight 1996).

Habitat Support for Fauna

None of the compensatory mitigation projects visited by the committee included design and evaluation criteria for animals. Animals are almost never manipulated or introduced into wetlands, in contrast to transplants of higher plants. Even when wetland assessments involve animals, the primary consideration is waterfowl and other birds or identifiable endangered/threatened species. Evaluations do not consider the constraint that most wetland animals are incapable of overland migration if terrestrial corridors are blocked by development, highway systems, or other situations not conducive to overland movement. Many wetland animal species are also dependent on the terrestrial habitat surrounding a prescribed wetland. The importance of considering migratory pathways and upland buffers in the design of a compensatory mitigation plan is discussed in greater detail in Chapter 3.

Soil Functions

Soil performs a number of important functions in a wetland that are usually overlooked in wetland restoration:

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- *Rooting medium.* Soil serves as a rooting medium for plants, providing the physical support for above-ground plant structures.
- *Germination medium.* Seed germination requires more specialized conditions than those required to sustain mature rooted plants. Germination of annuals, for example, is often promoted by a moist, temporarily exposed soil that is free of detritus.
- *Seed bank.* Seeds and rhizomes retained in the soil remain viable for months to years.
- *Source of water and nutrients for plants.* Soil is the site of water and nutrient uptake for rooted plants, even rooted plants that are submerged. The release of plant-available forms of nitrogen from unavailable organic forms stored in the soil (i.e., nitrogen mineralization) provides a constant source of nutrition to wetland plants.
- *Habitat for mycorrhizae and symbiotic bacteria.* Roots have complex relationships with soil fungi (mycorrhizae) and bacteria that enable and enhance nutrient uptake. Examples include nitrogen-fixing bacteria living symbiotically in root nodules of legumes and *Alnus* and vascular arbuscular mycorrhizae that associate with *Salix*. Some plants require the presence of specific mycorrhizal species to survive.
- *Water-quality functions.* The soil is the locus of most of the physical, chemical, and biological processes that give wetlands the ability to improve water quality. Sediment retention takes place at the soil surface. The chemical composition of the soil, such as the presence of iron and aluminum hydroxides, affects its ability to sorb phosphorus. Denitrifying bacteria dwell in the soil and depend on soil carbon as an energy source to support denitrification.
- *Habitat for soil macrofauna.* Soil-dwelling fauna sustain wading birds that probe the sediments of mud and sandflats with their long beaks. The role of soil-dwelling fauna in other types of wetlands is less well known.
- *Conduit for groundwater.* Soil permeability affects its ability to convey water. Dense, low-permeability soils may serve as aquacludes, causing water in wetlands to be perched above the regional water table. More permeable soils have higher hydraulic conductivities, allowing wetlands to have greater interaction with groundwater.
- *Source of contaminants.* Contaminants can be released from soils, particularly where the soil is landfill or has a prior history of industrial use. Soils that are high in heavy metals may release toxic forms, such as methylmercury and selenium, when creation of a wetland induces anaerobic conditions.

In wetland restoration and creation projects, soil is generally viewed as merely a rooting medium for the plants that are desired (the first function listed above). The soil that has developed in situ at a wetland creation

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site is often scraped off to attain a surface elevation that will allow the site to become flooded or intersect the water table. This soil-like material at depth usually has much different texture, structure, chemistry, and biota than the overlying material that was removed. It is nearly devoid of organic matter content, depauperate in nitrogen, may have been compacted by construction activities, and lacks mychorrizal and microbial populations important to plant establishment and water-quality functions. This material cannot provide the same functions as an intact wetland soil, and the plants that can successfully reproduce in such material may not be the desired ones.

The soil organic matter content of created wetlands has consistently been found to be less than that of reference wetlands (Craft et al. 1988, 1999; Gwin et al. 1990; Moy and Levin 1991; Zedler and Langis 1991, Kentula et al. 1992a; Bishel-Machung et al. 1996; Shaffer and Ernst 1999). In a multiple wetland study (n = 95), the average soil organic matter content of newly created wetlands did not change between 1987 and 1993, suggesting that there were no widespread improvements in wetland construction procedures to increase initial soil organic matter content (Shaffer and Ernst 1999). The vertical distribution of organic matter was also different in the profiles of created versus reference wetlands in Pennsylvania, with greater horizonation in the reference wetland soils (soil organic matter content significantly greater at depths of 5 centimeters (cm) than at 20 cm) than in the reference wetland soils (no significant difference in organic matter content at depths of 5 cm versus 20 cm). Although some researchers have reported increases in soil organic matter content in created wetlands over time (Lindau and Hossner 1981; Craft et al. 1988), others have found no significant relationship between soil organic matter content and project age (Bishel-Machung et al. 1996; Simenstad and Thom 1996; Shaffer and Ernst 1999). Low soil organic matter concentrations are associated with reduced levels of function, including poor establishment and growth of vegetation, poor habitat and food chain support for invertebrates and fish, and altered nutrient cycling (Shaffer and Ernst 1999).

Soil bulk density, which is the mass of soil per unit volume, has been found to be greater in created wetlands than in reference wetlands in at least two studies (Craft et al. 1991; Bishel-Machung et al. 1996). Soil texture also differed between created (n = 44) and reference wetlands (n = 20) in Pennsylvania: soils in created wetlands contained more sand and less clay at 20 cm, and reference wetlands were more silty than created wetlands were throughout the soil profile. Created wetlands may lack the redoximorphic features of low chroma and frequent redox depletions that are diagnostic of hydric soils (Confer and Niering 1992; Bishel-Machung et al. 1996).

Soil nutrient concentrations in created wetlands are generally lower

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than those in natural wetlands. The total nitrogen content of soil is usually positively related to organic matter content; hence, created wetland soils may contain less total nitrogen than do reference wetlands (Langis et al. 1991; Bishel-Machung et al. 1996). Wilson and Mitsch (1996) reported lower concentrations of phosphorus, potassium, calcium, and magnesium in constructed versus natural wetlands. Craft and colleagues (1991) reported lower concentrations of plant-available nutrients in a created salt marsh than in a reference salt marsh, which may have implications for plant primary productivity (Zedler and Langis 1991). Differences in the activity of soil microbes that control nutrient availability may also occur in created versus restored wetlands (Groffman et al. 1996).

Seeds and rhizomes retained in soil can remain viable for many years, germinating when conditions are right. Germination rates are a function of both environmental conditions (e.g., inundation, exposed mineral soils, and temperature) and seed conditions. For example, certain seeds have higher germination rates after a period of cold storage (stratification) or after abrasion of the seed covering (scarification; Thullen and Eberts 1995).

The presence of a seed bank can be either beneficial or detrimental. It is beneficial when desirable plant species naturally revegetate restored wetlands, reducing the cost and effort of planting programs (Mitsch and Wilson 1996; Mitsch et al. 1998). This approach is used most often in the restoration of preexisting wetlands (Galatowitsch and van der Valk 1996). Similarly, soil material with an intact seed bank can be transplanted from a wetland slated for destruction to a wetland creation site to promote revegetation (Brown and Bedford 1997). The presence of a seed bank can be detrimental, however; if it is dominated by weedy or invasive species, it would be viewed as undesirable. On several of the committee's site visits, the removal of soil was recommended for this reason. Planting of wetland vegetation in restored wetlands may not significantly influence the seed bank. Of 136 seed bank taxa found in soils from the shoreline of a reservoir constructed 6 years prior, only 10% were from species that had been planted during the restoration process (Collins and Wein 1995).

The experience at Hole-in-the-Donut in Everglades National Park (see Appendix B) illustrates the importance of restoring the substrate to achieve restoration of native plant communities. The soil that had been artificially created by rock plows during agricultural usage of the site provided substantially different conditions for plant growth than the oolitic limestone that supported the native sawgrass plant communities. Only by scraping off the artificially created soil was it possible to deter invasion by the invasive Brazilian pepper. When that was done, sawgrass regained its competitive advantage and was able to recolonize naturally without human intervention in the form of planting or weed control.

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FACTORS THAT CONTRIBUTE TO THE PERFORMANCE OF MITIGATION SITES

Wetland type and actions taken are two variables that contribute to the performance of mitigation sites. However, regardless of wetland type, the ability of a site to be restored depends in part on the degree to which it was degraded, and the degree to which it was degraded dictates, in large part, the actions that will be required to restore it (Zedler 1999). Degradation is a function of damages to both the watershed and the immediate site. Thus, restoration of a cattail marsh or open pond will be easier than a vernal pool, fen, or bog. And the creation or restoration of a cattail marsh or pond will be easiest at a site that is not too degraded in a watershed that is not too degraded. That is, the site should have a soil substrate (not bare rock as in a gravel pit); it should not be so contaminated that vegetation will not grow (as can be true where oil or pesticides have been spilled); and it should occur in a watershed that still retains the region's natural hydrology (e.g., groundwater should not be substantially depleted), as well as space and a topographic setting appropriate for a wetland to persist in perpetuity. Finally, the actions taken to restore or create a wetland should be appropriate to the type and degree of on-site degradation. If hydrology, soils, vegetation, and fauna have all been degraded, some attention to each of these classes of factors will likely be needed. Chapter 7 outlines appropriate guidelines for restoring or creating wetlands that are ecologically self-sustaining. The committee concludes that in a degraded wetland situation, many wetland functions are difficult to restore to their pre-disturbance condition. The ability to replace wetland functions depends on the particular function, the restoration-creation approach used, and the degree of degradation at the compensation site.

How Is the Likelihood of Achieving Wetland Restoration-Creation Goals Affected:

By the Hydrological Regime?

Wetlands are transition areas between water (wet 100% of the time) and uplands (seldom wet). In a qualitative sense, wetland hydrology has been defined by the National Research Council (NRC 1995) as recurrent, sustained inundation or saturation at or near the surface at a duration and frequency to support the development of diagnostic wetland features of hydric soils and hydrophytic vegetation. The NRC recognized that this definition is too broad to be applied directly in regulatory practice and so adopted quantitative criteria that focus on frequency, timing, and duration of inundation or soil saturation. In many cases, mitigation and resto-

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ration have required establishment of wetlands that are inundated or saturated more than 12% of the time ("seasonally inundated or saturated"; see Box 2-1). In reality, many impacted wetlands satisfy the saturation criteria closer to 5% of the time (Cole and Brooks 2000a). Thus, mitigated sites tend to be wetter than impacted sites (see Chapter 6 for more discussion on this consequence).

Our ability to restore or establish a particular hydrological regime is highly dependent on the desired wetland type. Wetlands that are surface-water-driven are, as a general rule, easier to establish than groundwater-dependent wetlands, but outcomes in both cases are highly dependent on landscape position. These wetland types are typically rainfall- or surface-runoff-driven, although some depressional areas, such as prairie potholes and the southeastern coastal plain, may rely on significant groundwater inflow. Hydrological regime is much more difficult to restore to an original state in highly modified hydrological systems (e.g., dams and levees). Hydrological restoration of prior converted cropland has been a generally successful process in part because the original conversion from wetland to agriculture resulted in relatively minor alteration of the natural hydrology. Wetland hydrology can often be restored to prior converted cropland by disrupting the artificial drainage system. Cooper et al. (1998) reported the restoration of hydrology to a Rocky Mountain fen where damage had been minor (i.e., installation of a drainage ditch). Monitoring data demonstrated that hydrology returned soon after the ditch was blocked. The return of natural water levels, however, might not restore soil chemistry or the ability to support calciphilic species, as discussed earlier. Important groundwater parameters are often inadequately characterized at reference and mitigation sites (Hunt 1996; Hunt et al. 1999). For riparian and riverine wetlands, the ability to restore hydroperiods depends on the degree to which streamflows have been modified (see Coyote Creek case study, Appendix B). Streams may have been channelized to convey storm water, and the need to protect upstream lands might preclude the restoration of natural flood pulses.

In conclusion, surface-water-dominated wetlands are easier to restore or create than groundwater-dominated wetlands, but achieving a natural hydrological regime is always a challenge. Hydrological regime is much more difficult to restore to the original state in highly modified hydrological systems (e.g., with dams and levees).

By Wetland Size?

Large natural wetlands are rarer, and their size imparts additional value to some functions (e.g., habitat for animals with large home ranges). Conversely, small isolated wetlands play a crucial role in the biodiversity

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of other wetland-dependent fauna, such as amphibians (Semlitsch 2000). There is no question that wetland size has an impact on water quality (Chescheir et al. 1991; Daniels and Gilliam 1996; Gilliam et al. 1997). Larger wetlands have a greater capacity to assimilate constituents in the inflow; however, the larger the wetland, the smaller the pollutant removal per unit area of wetland. Pollutant removal frequently follows a first-order decay curve with regard to residence time in the wetland. For a given input of water, the larger the wetland, the more pollutant will be removed. However, since each additional increment of land removes less pollutant than the previous increment, the removal per unit area of wetland is lower.

Shape may also influence the effectiveness of the wetland for pollutant removal. For functions such as water quality and nutrient retention, edge interface with stream or upland is probably more important than area. For example, pollutant removal by a 50-foot (ft)-wide wetland buffer beside a 100-ft-wide stream would remove far more contaminants than a 100-ft-wide buffer beside a 50-ft-wide stream. That would be true for water and pollutants entering the wetland from land upslope from the stream. However, that might not be true for water entering the wetland from stream bank overflow. However, for water-quality purposes, many small wetlands would be more effective than one large wetland covering the same area. The committee concludes that wetland size affects wetland functions. Thus, replacement area should be proportional to the area required to replace the functions lost.

By Wetland Place in the Landscape?

Riverine and slope wetlands are more difficult to restore or create than are depressional wetlands (Gwin et al. 1999; Shaffer et al. 1999). Position in the landscape affects a number of wetland functions, such as water quality (Johnston et al. 1990; Evans et al. 1993) and biodiversity (Poiani et al. 2000). Wetland place in the landscape is generally not considered a mitigation performance standard. The location of mitigation wetlands may be limited by the availability and cost of land.

Recently, much attention has been focused on the need to define hydrological equivalence in the landscape (Bedford 1996). Wetlands occur in a variety of physical settings, including coastal lowlands, topographic depressions, broad flats on interstream divides, the base of slopes, and topographic highs with little slope (Winter and Woo 1990). Location in the landscape influences geological characteristics such as slope; thickness and permeability of soils; and the composition, stratigraphy, and hydraulic properties of the underlying strata, all of which influence surface and subsurface flows of water. Quite simply, hydrology is the driv-

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ing force influencing wetland development, structure, function, and persistence. In conclusion, riverine and slope wetlands are more difficult to mitigate than are depressional wetlands.

By the Ecoregion in Which a Wetland Occurs?

Ecoregions are quite diverse, and it was difficult for the committee to generalize about the effect of ecoregion on the likelihood of achieving wetland restoration-creation goals. Created wetlands in areas of variable precipitation, one factor that distinguishes ecoregions, may not meet the jurisdictional definition of a wetland every year. Such areas are more likely to occur in arid ecoregions, although periods of drought and temporal variations in precipitation can occur in any ecoregion. Innate temporal variability of wetlands due to climatic variation is not the same as mitigation failure.

The committee concludes that created wetlands in regions of variable precipitation may not meet the jurisdictional definition of a wetland every year, increasing the risk of noncompliance with performance standards. Thus, the mitigation design should recognize and accommodate hydrological variability and extremes caused by climate. Wetland restoration and enhancement should be preferred over creation in such areas.

By the Kinds of Plants Present?

An important general consideration of wetland design is whether plant material is going to be allowed to develop naturally from some initial seeding and planting or whether continuous horticultural selection for desired plants will be imposed. To develop a wetland that will ultimately require low maintenance, natural successional processes need to be allowed to proceed. For forested wetlands, an initial period of invasion by undesirable species might be temporary if proper hydrological conditions are imposed and if trees shade out early invaders. One strategy is to introduce, by seeding and planting, many of the available species to allow natural processes to sort out the species and communities over time. Selective weeding may be necessary in the beginning or throughout the life of the wetland if aggressive exotic vegetation persists. Preferably, the system can sustain itself through its own successional patterns. Otherwise, labor-intensive management, which is never desirable in a compensatory mitigation wetland, will be needed. In some cases, survival of specific plantings is used to evaluate compliance. Wetlands that are readily invaded by exotics or prove to support undesirable monotypes over long periods require greater attention to planting and an establishment-phase exotic control program (see Boxes 2-2 and 2-3).

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BOX 2-2
Seeding Versus Natural Recruitment

Reinartz and Warne (1993) found that early introduction of a diversity of wetland plants may enhance the long-term diversity of vegetation in created wetlands. The study examined the natural colonization of plants in 11 created wetlands in southeastern Wisconsin. The wetlands under study were small isolated, depressional wetlands. A 2-year sampling program was conducted for the created wetlands, aged 1-3 years. Colonized wetlands were compared with five seeded wetlands where 22 species were introduced. The diversity and richness of plants in the colonized wetlands increased with age, size, and proximity to the nearest wetland source. In the colonized sites, *Typha* spp. comprised 18% of the vegetation for 1-year wetlands and 50% for 3-year wetlands, with the possibility of monocultures of *Typha* spp. developing over time in colonized wetlands. The seeded wetlands had a high species diversity and richness after 2 years. *Typha* cover in these seeded sites was lower than that in the colonized sites after 2 years.

In conclusion, wetland mitigation designs should include plantings (e.g., sedges over cattails). Unless actively controlled at the outset, exotic and weedy plant species often dominate restoration sites. Species richness is often low in created wetlands.

By the Kinds of Animals Present?

Natural freshwater wetlands support among the highest levels of regional species' diversity and population densities of fauna in North

BOX 2-3
Does Wetland Planting Help Self-Design?

In a multiyear study of the effect of plant introduction on ecosystem function, researchers at the Olentangy River Wetland Research Park in Ohio found that a planted wetland and an unplanted wetland converged in most functions (eight biological measures; eight biophysicochemical measures) in 3 years (Mitsch et al. 1998). Continued studies showed a persistence of the planted vegetation in that basin but dominance by *Typha* in the naturally colonizing basin, with differences in function between the two basins. The planted wetland had more plant communities but had 50% lower net primary productivity, higher summer water temperature, and lower macroinvertebrate diversity than did the naturally occurring wetland 6 years after planting (Mitsch et al. 1999).

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America. Several biological and landscape considerations are critical to the sustainability of such species, and a variety of factors must be considered in attempting compensatory mitigation for loss of animal-support functions. Created wetlands are generally not designed to meet the needs of animals found in the impacted wetland; hence, animal species' richness is often low in mitigation sites.

Amphibians are perhaps the best-studied group of organisms from the standpoint of wetland dependency both in terms of association with a particular wetland and the landscape pattern of wetland connectivity. The importance of amphibians as a major component of wetland biodiversity, the significance and necessity of peripheral terrestrial habitat to their existence, and the requirement for wetland interconnectivity in a landscape have been documented for a variety of species and regions in North America (Berven and Gill 1983; Berven and Grudzien 1990; Berven 1990; Pechmann et al. 1991; Dodd 1992, 1993, 1995; Gibbs 1993; Semlitsch et al. 1996; Snodgrass et al. 1999; Madison 1997; Semlitsch 1998, 2000; Semlitsch and Bodie 1998; Lamoureux and Madison 1999). Biological dynamics of animal populations are discussed in Chapter 3.

The committee concludes that for compensatory mitigation of a wetland to be effective for all affected fauna, the biological dynamics must be evaluated in terms of the populations present and the ecological requirements of the species, which include metapopulation aspects that are affected by the relationship of the wetland to other wetlands in the local system.

By Time?

After more than 2 decades of compensatory mitigation, there are now thousands of hectares of restored and created wetlands in the United States. Yet only a few studies have analyzed how various ecosystem components have changed over time, and even fewer describe ecological performance over more than 5 years. Knowing how rapidly an ecosystem matures, or fails to mature, is important for learning how to improve future projects.

A 3- to 5-year monitoring period is a common permit requirement. Five years may be enough time for herbaceous plant cover to reach a peak (see Figure 2-1), but trees obviously take longer than herbaceous plants to reach peak biomass. Biomass, however, may not be equivalent between the restored or created wetland and the natural reference ecosystems. For example, Brown and Veneman (1998) analyzed 68 paired mitigation sites in Massachusetts and, using criteria for establishment for equivalency, reported that plant communities were not equivalent to reference systems after 13 years.

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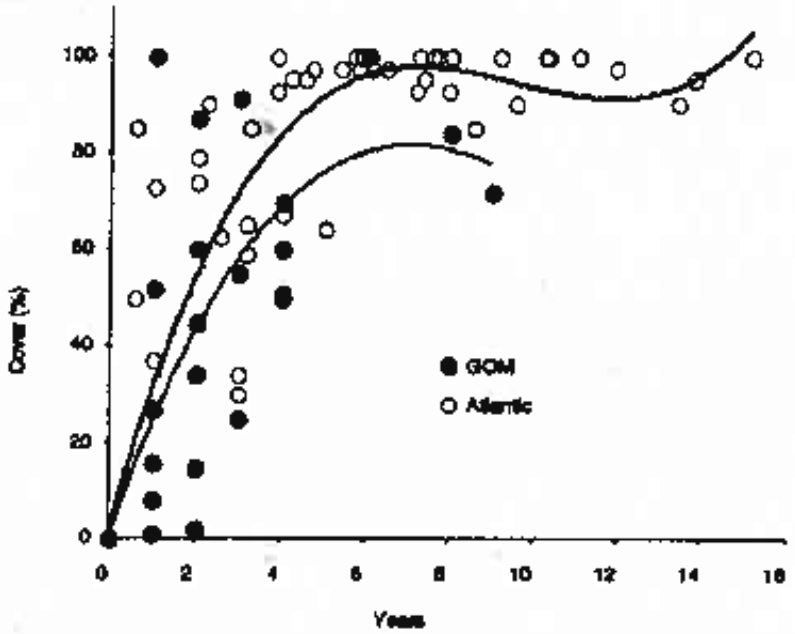

FIGURE 2-1 Percent plant cover on created or restored coastal wetlands on the Atlantic and Gulf of Mexico (CCB4) coasts. A second-degree polynomial fit of the data is shown. Note that 100% cover was achieved in most of the Atlantic wetlands within 5 years. SOURCE: Adapted from Matthews and Minello (1994).

The animal components of communities may develop at quite different rates than do plants at mitigation sites. Mobile species may migrate in and colonize with the first floodwaters. Even with fish, however, simple analyses of population structure indicate that the trajectory from colonization to stability is not equal or smooth among population types. For one study, Rakocinski (1991) surveyed a created salt marsh constructed as a mitigation site in North Carolina. Although this study was only for a single site, the findings are important. Rakocinski found that the number of species captured with one piece of gear (trawl) became more like the number of species in the reference marsh within 20 months and after 42 months for fish collected with shallower-water gear (Wigener Ring). Neither the number of organisms captured with each gear type nor the number of similar species was stable after 4 years. The equilibrium position of various fisheries parameters was not reached within 4 years of the man-

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TABLE 2-1 Summary of Results from Study of a Created Salt Marsh Constructed as a Mitigation Site in North Carolina (1991)

Comparison of Finfish in a Constructed Wetland Compared with Two Reference Sites	Months to Reach Reference Site Conditions	Trend After Equivalency First Reached
Number of organisms		
Trawl	8	Falling
Wegener Ring	28	Rising
Number of species		
Trawl	18	Stable
Wegener Ring	43	?
Species similarity		
Trawl	0-12	Declining
Wegener Ring	0-12	Declining

SOURCE: Adapted from Rulifson (1991).

agement action but then continued to change. In other words, the equilibrium was obtained but not equivalency (see Table 2-1). Zedler (1990) noted that the average number of individual fish species and the average number of individuals caught at a single reference site and a wetland creation project in Southern California were reached within 5 years. However, the similarities of fish and benthic species between the reference site and mitigation sites were only 58% and 54%, respectively.

Results from a variety of trajectory analyses of soil, plant, and animal communities for mitigation sites are given in Appendix C and summarized in Table 2-2. Plant cover and biomass for some marshes may reach

TABLE 2-2 Time Toward Equivalency for Soil, Plant, and Animal Components in Wetland Restoration Projects Compared with That of Natural Reference Wetlands

Structural Component	Number of Sites	Range (years)
Soils	17	>3 to >30
Wetland plants		
Cover or biomass	7	3 to >20, or never
Species composition	3	>5 to >10, or never
Below-ground biomass	2	10+, or never
Fish and fisheries	9	>2 to 10
Marsh fisheries	14	1 to >17, or never
Birds	7	<3 to >15, or never

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an equilibrium in 3 years, as discussed above, but species composition may take 10 years or longer to stabilize, if at all. Trees, obviously, will not reach an equilibrium until at least the lifetime of the tree. The trajectories for soils, plants, and animals are not the same (e.g., Zedler and Callaway 1999; see Figure 2-2). In contrast to herbaceous vegetation, soil development may be quite slow (Craft et al. 1999), taking from 3 years to 30 years to reach equilibrium, if equilibrium is realizable. The fish and bird components may reach equivalency in as little as 2 years for some species. The trajectory may never reach an equilibrium or the equilibrium conditions may not equal those at the reference sites. However, because there has

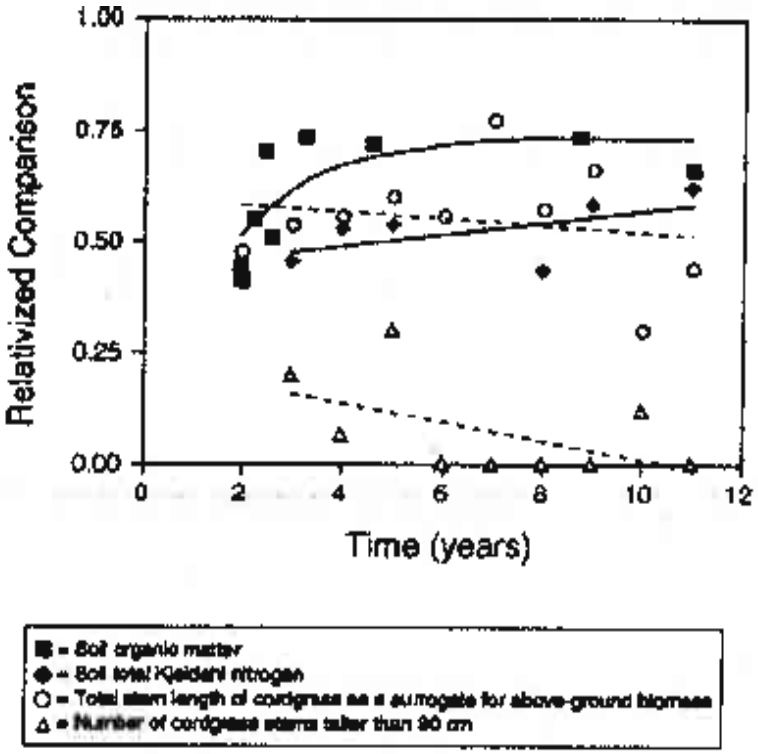

FIGURE 2-2 Long-term data for salt marshes constructed in San Diego Bay. Values are in relation to the adjacent natural salt marsh. The site was required to provide self-sustaining tall cordgrass stems for nesting by an endangered clapper rail. SOURCE: Adapted from Zedler and Callaway (1999).

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been virtually no research that compares the relative similarity of undisturbed reference sites, it is hard to make firm conclusions on this point. In their analysis of reference site selection, White and Walker (1997) found that there is so much variation on a regional scale that there may never be a perfect match for a site to be restored. They propose that a conceptual restoration model should be one of interpolation among multiple sites and sources of information, including temporally. Finally, Neiring (in Kusler and Kentula 1990) posits that the end point of the successional process cannot be predicted because it is not an orderly process; that is, we cannot predict what exact species will be at the "end" of the process. In summary, it appears that there is no general trajectory for the development of wetland ecosystems or individual components (structural or functional attributes) of a single watershed.

The significance of these results is not that equivalency among reference and newly managed environments is not reached or that mitigation efforts should not be done. These results demonstrate that (1) ecological equivalency may *not* be reached within a few months or for several years or even decades, depending on the attribute that is of interest; (2) the ecosystem does not move smoothly to an equilibrium or at the same rate for all components; and (3) some components, including ones identified as important in permits currently being issued, may *never* reach equivalency with the natural reference wetland. An obvious conclusion from these results, besides the general paucity of scientific analyses, is that the generally observed 5-year limit on monitoring is insufficient when evaluating whether a site has achieved parity with a reference system. Further, the amount of mitigation required should be based on the amount needed to fully offset the permitted wetland losses. To accomplish that, mitigation ratios (area of mitigation to area lost) will often need to be increased to achieve functional equivalency, rather than simply matching wetland area.

The above review of wetland restoration and creation outcomes has not differentiated projects on the basis of their starting conditions. Some of the projects discussed occurred at sites where damages were relatively minor (e.g., Cooper et al. 1998); others were at highly altered sites with artificial (dredge spoil) substrate (e.g., Zedler and Callaway 1999). As discussed earlier, it is very likely that the ability to achieve desired outcomes is a function of the degree of site degradation. Degradation, in itself, is a complex concept, involving not only the local site but also its watershed. Zedler (1999) suggests that generalizations are not easily drawn among restoration efforts involving sites with different degrees or types of degradation, suggesting instead that ecologists seek predictability of outcomes in a matrix of situations, where degree of degradation is one axis and the type of restoration effort is the other. The general pattern

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is likely to be that outcomes are more predictable where damages are minimal and restoration efforts most intensive. Badly damaged wetlands and wetland creation sites are less likely to match reference wetlands, especially if restoration efforts are minimal.

In conclusion, the literature and long-term trajectories reported therein suggest that wetland restoration and creation sites do not often achieve functional equivalency with reference sites within 5 years; indeed, up to 20 years may be needed for functional attributes to be determined or assessed correctly.

RECOMMENDATIONS

On the basis of its evaluation of wetland structure and function, the committee makes the following recommendations for compensatory mitigation:

1. Avoidance is strongly recommended for wetlands that are difficult or impossible to restore, such as fens or bogs.
2. The science and technology of wetland restoration and creation need to be based on a broader range of studies, involving sites that differ in degree of degradation, restoration efforts made, and regional variations. Predictability of outcomes should then improve.
3. All mitigation wetlands should become self-sustaining. Proper placement in the landscape to establish hydrogeological equivalence is inherent to wetland sustainability.
4. Hydrological variability should be incorporated into wetland mitigation design and evaluation. Except for open-water wetlands, static water levels are not normal. Because of climatic variability, it should be recognized that many wetland types do not satisfy jurisdictional criteria every year. Hydrological functionality should be based on comparisons to reference sites during the same time period.
5. Because a particular floristic assemblage might not provide the functions lost, both restoration of community structure (e.g., plant cover and composition) and restoration of wetland functions should be considered in setting goals and assessing outcomes. Relationships between structure and function should be better known.
6. The biological dynamics should be evaluated in terms of the populations present in reference models for the region and the ecological requirements of those species.
7. Mitigation projects should be planned with and measured by a broader set of wetland functions than are currently employed.

EXHIBIT 11

SCIENCE AND REGULATION

Mountaintop Mining Consequences

Damage to ecosystems and threats to human health and the lack of effective mitigation require new approaches to mining regulation.

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There has been a global, 30-year increase in surface mining (*1*), which is now the dominant driver of land-use change in the central Appalachian ecoregion of the United States (*2*). One major form of such mining, mountaintop mining with valley fills (MTM/VF) (*3*), is widespread throughout eastern Kentucky, West Virginia (WV), and southwestern Virginia. Upper elevation forests are cleared and stripped of topsoil, and explosives are used to break up rocks to access buried coal (fig. S1). Excess rock (mine "spoil") is pushed into adjacent valleys, where it buries existing streams.

Despite much debate in the United States (*4*), surprisingly little attention has been given to the growing scientific evidence of the negative impacts of MTM/VF. Our analyses of current peer-reviewed studies and of new water-quality data from WV streams revealed serious environmental impacts that mitigation practices cannot successfully address. Published studies also show a high potential for human health impacts.

Ecological Losses, Downstream Impacts

The extensive tracts of deciduous forests destroyed by MTM/VF support some of the highest biodiversity in North America, including several endangered species. Burial of headwater streams by valley fills causes permanent loss of ecosystems that play critical roles in ecological processes such as nutrient cycling and production of organic matter for downstream food webs; these small Appalachian streams also support abundant aquatic organisms, including many endemic species (*5*). Many studies show that when more than 5 to 10% of a watershed's area is affected by anthropogenic activities, stream biodiversity and water quality suffer (*6*, *7*). Multiple watersheds in WV

already have more than 10% of their total area disturbed by surface mining (table S1).

Hydrologic flow paths in Appalachian forests are predominantly through permeable soil layers. However, in mined sites, removal of vegetation, alterations in topography, loss of topsoil, and soil compaction from use of heavy machinery reduce infiltration capacity and promote runoff by overland flow (*8*). This leads to greater storm runoff and increased frequency and magnitude of downstream flooding (*9*, *10*).

Water emerges from the base of valley fills containing a variety of solutes toxic or damaging to biota (*11*). Declines in stream biodiversity have been linked to the level of mining disturbance in WV watersheds (*12*). Below valley fills in the central Appalachians, streams are characterized by increases in pH, electrical conductivity, and total dissolved solids due to elevated concentrations of sulfate (SO_4), calcium, magnesium, and bicarbonate ions (*13*). The ions are released as coal-generated sulfuric acid weathers carbonate rocks. Stream water SO_4 concentrations are closely linked to the extent of mining in these watersheds (*11*, *14*). We found that significant linear increases in the concentrations of metals, as well as decreases in multiple measures of biological health, were associated with increases in stream water SO_4 in streams below mined sites (see the chart on page 149). Recovery of biodiversity in mining waste-impacted streams has not been documented, and SO_4 pollution is known to persist long after mining ceases (*14*).

Conductivity, and concentrations of SO_4 and other pollutants associated with mine runoff, can directly cause environmental degradation, including disruption of water and ion balance in aquatic biota (*12*). Elevated SO_4 can exacerbate nutrient pollution of downstream rivers and reservoirs by increasing nitrogen and phosphorus availability through internal eutrophication (*15*, *16*). Elevated SO_4 can also increase microbial production of hydrogen sulfide, a toxin for many aquatic plants and organisms (*17*). Mn, Fe, Al, and Se can become further concentrated in stream sediments, and Se bioaccumulates in organisms (*11*) (figs. S1 and S2).

A survey of 78 MTM/VF streams found that 73 had Se water concentrations greater than the 2.0 μg/liter threshold for toxic bioaccumulation (*18*). Se levels exceed this in many WV streams (see the chart on page 149). In some freshwater food webs, Se has bioaccumulated to four times the toxic level; this can cause teratogenic deformities in larval fish (fig. S2) (*19*), leave fish with Se concentrations above the threshold for reproductive failure (4 ppm), and expose birds to reproductive failure when they eat fish with Se >7 ppm (*19*, *20*). Biota may be exposed to concentrations higher than in the water since many feed on streambed algae that can bioconcentrate Se as much as 800 to 2000 times that in water concentrations (*21*).

Potential for Human Health Impacts

Even after mine-site reclamation (attempts to return a site to premined conditions), groundwater samples from domestic supply wells have higher levels of mine-derived chemical constituents than well water from unmined areas (*22*). Human health impacts may come from contact with streams or exposure to airborne toxins and dust. State advisories are in effect for excessive human consumption of Se in fish from MTM/VF affected waters. Elevated levels of airborne, hazardous dust have been documented around surface mining operations (*23*). Adult hospitalizations for chronic pulmonary disorders and hypertension are elevated as a function of county-level coal production, as are rates of mortality; lung cancer; and chronic heart, lung, and kidney disease (*24*). Health problems are for women and men, so effects are not simply a result of direct occupational exposure of predominantly male coal miners (*24*).

Mitigation Effects

Reclamation of MTM/VF sites historically has involved planting a few grass and herb species (*20*, *25*). Compared with unmined

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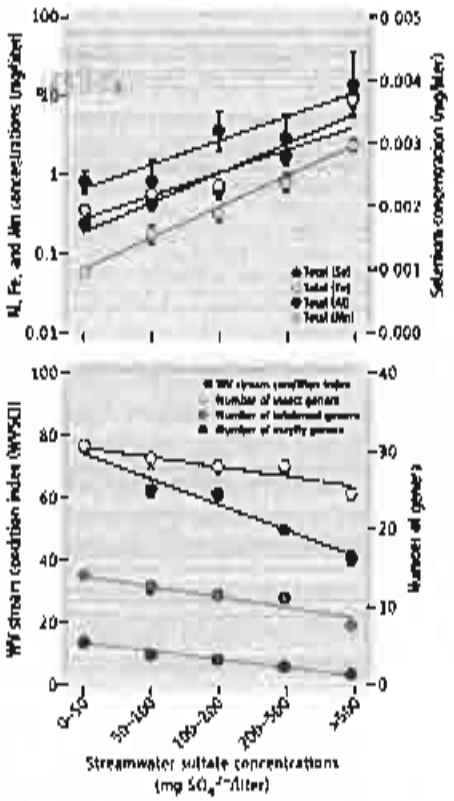

Mining effects on stream chemistry and biota. Sulfate concentrations reflect amount of mining in watershed. (**Top**) Average concentrations of manganese, iron, aluminum, and selenium. (**Bottom**) Stream invertebrate community metrics in relation to sulfate concentrations for 1058 WV streams (methods in table S2). Regressions all statistically significant (table S3).

sites, reclaimed soils characteristically have higher bulk density, lower organic content, low water-infiltration rates, and low nutrient content (*8*, *25*). Many reclaimed areas show little or no regrowth of woody vegetation and minimal carbon (C) storage even after 15 years (*26*). Decreased forest productivity may be related to the type of surface material (e.g., brown versus gray sandstone) used in the reclamation (*27*). In reclaimed forests, projected C sequestration after 60 years is only about 77% of that in undisturbed vegetation in the same region (*28*). Mined areas planted to grassland sequester much less. Since reclamation areas encompass >15% of the land surface in some regions (*29*) (table S1), significant potential for terrestrial C storage is lost.

Mitigation plans generally propose creation of intermittently flowing streams on mining sites and enhancement of streams offsite. Stream creation typically involves building channels with morphologies similar to unaffected streams; however, because they are on or near valley fills, the surrounding topography, vegetation, soils, hydrology, and water chemistry are fundamentally altered from the premining state. U.S. rules have considered stream creation a valid form of mitigation while acknowledging the lack of science documenting its efficacy (*30*). Senior officials of the U.S. Army Corps of Engineers (ACOE) have testified that they do not know of a successful stream creation project in conjunction with MTM/VF (*31*).

A Failure of Policy and Enforcement

The U.S. Clean Water Act and its implementing regulations state that burying streams with materials discharged from mining should be avoided. Mitigation must render nonsignificant the impacts that mining activities have on the structure and function of aquatic ecosystems. The Surface Mining Control and Reclamation Act imposes requirements to minimize impacts on the land and on natural channels, such as requiring that water discharged from mines will not degrade stream water quality below established standards.

Yet mine-related contaminants persist in streams well below valley fills, forests are destroyed, headwater streams are lost, and biodiversity is reduced; all of these demonstrate that MTM/VF causes significant environmental damage despite regulatory requirements to minimize impacts. Current mitigation strategies are meant to compensate for lost stream habitat and functions but do not; water-quality degradation caused by mining activities is neither prevented nor corrected during reclamation or mitigation.

Clearly, current attempts to regulate MTM/VF practices are inadequate. Mining permits are being issued despite the preponderance of scientific evidence that impacts are pervasive and irreversible and that mitigation cannot compensate for losses. Considering environmental impacts of MTM/VF, in combination with evidence that the health of people living in surface-mining regions of the central Appalachians is compromised by mining activities, we conclude that MTM/VF permits should not be granted unless new methods can be subjected to rigorous peer review and shown to remedy these problems. Regulators should no longer ignore rigorous science. The United States should take leadership on these issues, particularly since surface mining in many developing countries is expected to grow extensively (*32*).

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EXHIBIT 12

Report on Chuitna Coal Project of PacRim Coal

prepared by

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March 16, 2009

This report is in relation to a proposed coal mining project that will impact the Chuitna River watershed which is situated near Anchorage, Alaska and to the west side of Cook Inlet. The coal is located in a 20,571-acre lease tract. The project will involve putting in access roads to the mining site, building housing, adding an airstrip, power facilities, logistics support facilities, and various other mining-related infrastructure (e.g., transport conveyors). The watershed is ~150 square miles but the area to be impacted is much larger due to all the infrastructure and access needs for mining.

As context for my comments, I am a Professor at the University of Maryland where I oversee a scientific research laboratory with approximately 120 staff, 20 buildings, and a research fleet (see attached Curriculum vitae). I have over 25 years of experience in research and teaching on coastal ecosystems, watershed science, and stream ecology and restoration. Past work includes leading a large team of scientist in developing the first national database on river restoration in the U.S., co-authoring a book on *The Foundations of Restoration Ecology* and serving as an expert advisor on the design of multiple stream and river restoration projects.

I have reviewed a number of documents including the 2007 Fish and Wildlife Protection Plan (Part D7), the Protection of the Hydrological Balance (Party D12), the Hydrology Component Baseline Report, the Annual Water Management Plans (Addendum D12-A), the Baseline Report for Vegetation and Wetlands, a Wetlands Functional Assessment, and portions of the 1990 Final Environmental Impact Statement by the EPA and other related documents including some appendices.

I begin with comments on the impacts of the mining activities; however, my report focuses primarily on concerns with the mitigation that is for the streams.

Ecological impacts of the project

The Chuitna River watershed is characterized largely by tundra vegetative cover, spruce-birch forest (almost 50% of the mine site), alder scrub, and various fen and wetland species. Many of the plants and habitat types are very sensitive ecologically and yet will be destroyed or degraded severely. The impacts of the mining activities include destruction of forest, wetlands, habitat for wildlife, habitat for aquatic species, and the entire loss of 17.4 km of streams that support healthy populations of invertebrates and fish, including many highly valued salmon and other game fish. Tributaries of the Chuitna River that will be impacted include in particular, the headwater stream called "2003 Creek" (sometimes called Middle Creek) which drains the mine site; other tributaries likely to be impacted include "Lone Creek" and "2004 Creek".

Critical habitat. The areas to be impacted are in pristine regions of the watershed and while one could argue that all of the habitat types are critical ecologically, there are three that are unusually so: tundra, wetlands, and headwater streams (Elmqvist et al. 2003)

Tundra ecosystems are considered among the most fragile ecosystems on earth (Reynolds and Tenhunen 1996). They are extremely sensitive to disturbances (even dust from nearby roads), slow to recover (Myers-Smith et al. 2006) and a huge fraction of the carbon in tundra ecosystem is stored in soils (Mack et al. 2004) which will be severely disturbed by the proposed mining activity. Yet the ability to restore the tundra regions and peatlands after the Chuitna Coal Project is complete is highly questionable. Peatland restoration has not been studied very extensively but current work indicates that only under the following conditions are restoration efforts likely to be successful: some remnant vegetation, a seed bank (in the soil), and connection to other healthy peatlands (Gorham and Rochefort 2003). For much of the region that is impacted by the mining, these conditions will not be met.

Wetland areas are also well known as ecologically critical ecosystems. The reports from PacRim indicate that approximately 4,000 acres of wetlands are in the Chuitna Coal Mine mapping area and there are at least 7 plant species designated as 'rare' by the Alaska Natural Heritage Program. Wetland types include riverine lowlands, bogs, fens, and areas around ponds. The project is in a broad region of Alaska well known for its ecologically important wetlands (Tiner 2003). These wetlands have high rates of primary production and store large amounts of organic matter, both of which are critical to the overall food web of the region as well as to long term carbon storage (Chimner et al. 2002). Wetlands such as these support both photosynthetically-based food webs and food webs dependent on detritus (decaying organic matter) and are home to a very diverse community of permanent and migratory wildlife (Mitsch and Gosselink 2007). Such wetlands are also critical sites for water storage and groundwater recharge (Brauman et al. 2007). Finally, wetland ecosystems play pivotal roles in biogeochemical processes including nutrient transformations, denitrification, removal of some contaminants from the water, and decomposition (Palmer and Richardson 2009, see Table 1). There is an extensive literature on wetland restoration indicating that hydrological flow paths are essential to success and the Chuitna mining project will fundamentally alter these paths. Further, results from wetland creation efforts suggest that while mitigation projects may meet compliance requirements, full ecological or functional success is low or unknown for most projects (Ambrose et al. 2007; Euliss et al. 2008; Mathews and Endress 2008). A 2008 review prepared for Congress stated: "Both scientists and policymakers debate whether it is possible to restore or create wetlands with ecological and other functions equivalent to or better than those of natural wetlands that have been lost over time" (Copeland and Zinn 2008).

Headwater streams are those smallest of all tributaries that reflect 'where rivers are born' – that is, they feed the complex network of larger and larger downstream tributaries. Their position within watersheds (they impact all downstream waters), their high rates of key biogeochemical processes, and the high levels of biodiversity they support have been emphasized in a great deal of scientific research (Lowe and Likens 2005; Meyer et al. 2007). Headwater streams such as those that will be destroyed or impacted by watershed disturbance during mining may be small in size, but they provide habitats for a rich array of species, which enhances the biological diversity of the entire river system. This is discussed later but briefly: they provide a source of food and colonizing flora and fauna to downstream waters, they are important spawning and nursery grounds for fish and insects that live in larger rivers the rest of their lives, and they provide a refuge from predators and changes

in temperature for some species. Their role in biogeochemical processing and particularly nitrogen dynamics is disproportionately related to their size (Peterson et al. 2001). As I discuss later, the potential for headwater streams to be restored varies with the amount of degradation; however, there is no scientific evidence that streams that are mined through in the manner PacRim proposed can ever be restored ecologically.

Biodiversity loss and functional consequences. The loss of 17.4 km of streams will remove habitat that is essential to the life cycle of salmonids and other groups of fish that reside all or part of the year in the Chuitna tributaries. Mining through the streams to a depth of > 300 feet will lead to severe and permanent environmental impacts to the existing channel and living resources. Attempts will be made to re-locate valued fish but survival is not certain. Mobile animals like bear, moose, and birds may be able to move to other areas when their local habitat is destroyed although some mortality is inevitable. Flora and smaller fauna within the streams to be mined will be killed including amphibians, invertebrates, algae, and microbial communities. Previous reports for the Chuitna watershed indicate that invertebrates include not only taxa such as chironomids and simulid blackflies (Diptera ns) but sensitive insect taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera. Thus, biodiversity losses will be very significant in the streams that are mined and in downstream reaches which are impacted by the upstream disturbances.

Biodiversity loss in the Chuitna watershed is of great concern not only because valued species will be lost but because biodiversity is integral to ecosystem function. Biodiversity loss is often associated with a decline if one or more ecosystem functions (Hooper et al., 2005; Cardinale et al., 2006) and since different species contribute differentially to different functions, the maintenance of multi-functional ecosystems requires maintenance of high species diversity (Hector & Bagchi, 2007). Because diverse ecosystems are typically associated with higher functional and response diversity (different species vary in their vulnerability and response to change), ecosystems with many species exhibit more resilience in the face of environmental changes (Elmqvist et al. 2003). Ecological resilience is the ability to resist perturbations or recover from disturbances such as unusual climate events (storms, droughts) or even anthropogenic disturbances (e.g., global warming, tree harvest). Once biodiversity is degraded, ecosystems are more vulnerable to collapse due to higher temporal and spatial fluctuations in species and performance; in short, species diversity

contributes to the stability of biotic communities and ecosystem function (Naeem et al. 1994; Tilman et al. 1996).

Deficiencies of the Mitigation Plan

The proposed mitigation includes the construction of artificial off-channel spawning and rearing habitat, off-site mitigation options (culvert improvements, bridge repairs, erosion control on the Theodore River, lake "enhancement", removal of a dam on Fish Creek), and stream "reclamation". I focus my comments on the latter since it is the only activity that directly addresses compensation for the destruction of the 17.5 km of Stream 2003 and associate tributaries.

The PacRim *Fish and Wildlife Protection Plan* describes the objective of fully reconstructing an "undisturbed" channel and floodplain to "assure pre-mined ecological functions and values are restored" (page 15). A natural channel design (hereafter, NCD) or Rosgen approach is proposed. They will attempt to mimic the morphological attributes (channel dimension, pattern, profile) of a 'reference reach' that is selected from the existing, pre-mined channel. Their goal is to "replicate the pre-mine channel geometry..." (page 24). They have extensive data from geomorphic surveys, streambed particle size measurements, and habitat surveys and they have classified existing channels based on the Rosgen classification scheme. Stream 2003 is apparently not gauged at present and so they will estimate hydraulic parameters needed for the design. It also appears they do not have direct measurements of bedload or tractive force in existing channels but indicate they will get "direct sediment discharge collections across a range of flows" (page 24) in order to develop a sediment rating curve. Descriptions of the ecological aspects of the reclamation are extremely limited and mostly deal with riparian vegetation.

1. <u>Failure to directly assess ecosystem functions.</u> The mitigation plan does not include direct assessment of the ecological functions that will be lost when the streams are destroyed by the mining activities. Healthy streams are living, functional systems and the most essential ecological functions include: the purification of water, the removal of excessive levels of nutrients and sediments before they reach downstream waters, the processing of organic material (decomposition or biological utilization), and primary and secondary productivity (growth of photosynthetic organisms and

consumers) (Fischenich 2006, Allan and Castillo 2007). The most common stream functions are, shown below in Table 1.

These functions are supported by ecological processes including: the processing of nutrients at the same rate and form as unimpacted streams, the decomposition of organic matter at rates typical of nearby unimpacted streams, and, microbial, primary and secondary production the same as nearby healthy streams (Palmer et al. 1997a; Naiman et al. 2006; Palmer and Richardson 2007). These processes must be measured in order to determine how and whether they may be brought back to the right levels and direction through restoration. No data or measurement plans for these processes are provided in the mitigation plan despite abundant scientific studies outlining how to make and interpret such measurements (e.g., Peterson et al. 2001; Gessner and Chauvet 2002; Hall and Tank 2003) and how such measurements can be used to evaluate the success of a restoration project (Buckveckas et al. 2007; Roberts et al. 2007).

Use of well-accepted methods for measuring ecological functions (e.g., see Hauer and Lamberti 1996, 2006) is important because ecological functions evaluate dynamic properties of ecosystems that underlie an ecosystem's ability to provide vital goods and services (Gessner and Chauvet 2002, Falk et al. 2006, Fischenich 2006, Palmer and Richardson 2007). The units of an ecological function are a process rate and direction (Table 1). Functions reflect system performance (http://www.epa.gov/owow/functions.html) and their measurement requires quantification of ecological processes such as primary production or nutrient uptake (Hauer and Lamberti 1996, 2006). This should be reflected in the mitigation plan if the plan is to mitigate functions that are lost due to the mining through of streams. Functional measures have been used to compare degraded vs. restored vs. reference streams[1] and have been shown to be quite sensitive to degradation and restoration.

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TABLE 1 ECOLOGICAL FUNCTIONS OF HEADWATER STREAMS

Ecosystem function	Ecological Process that supports this	Measurements required	Without it what happens
Water Purification a) Nutrient Processing	Biological uptake and transformation of nitrogen, phosphorus	Direct measures of **rates** of transformation of nutrients; for example: microbial denitrification, conversion of nitrate to N_2	Excess nutrients can build up in the water making it unsuitable for drinking or to support life
Water Purification b) Processing of contaminants	Biological removal of materials such as excess sediments (removed by riparian plants for example) or such as toxins (some taken up by plants or processes by microbes thereby removing them from the water)	Direct measures of contaminant flux (e.g., the movement of sediment into and down streams). This is a **rate.**	Toxic contaminants kill biota; excess sediments smother invertebrates (kill them), foul the gills of fish (kill them), etc; water not potable
Decomposition of organic matter (organic matter processing)	The biological (mostly by microbes and fungi) degradation of organic matter (could be leaf material or other input such as sweater or organic wastes)	Decomposition is measured as a **rate.** Usually expressed as the slope of a line showing weight loss over time of organic matter heated to high temperatures to convert the particulate carbon to gas (CO2)	Without this, excess organic material builds up in streams, leading to low oxygen levels which leads to death of invertebrates and fish and the water is not something anyone would want to drink
Production (Primary = algae & aquatic plant; Secondary = growth of organisms like insects, fish, etc	Measured as a rate of new plant or animal tissue produced over time	Primary - measure the **rate** of photosynthesis in the stream; for secondary, you measure growth rate of organisms	Primary production supports the food web; secondary production (fish) we often eat or it (inverts) supports fish.
Temperature Regulation	Water temperature is "buffered" (i.e., does not change dramatically) if there is sufficient infiltration in the watershed & riparian zone (due to vegetation) AND shading of the stream by riparian vegetation keeps the water cool.	Measure the **rate** of change in water temperature as air temperature changes or as increases in discharge occur.	If water infiltration or shading reduced (due to clearing of vegetation), the stream water heats up beyond what biota are capable of tolerating (due to high sunlight reaching the stream and an increase in overland runoff)
Flood Mediation/Control	Slowing of flow from land to streams so flood frequency and magnitude reduced; intact flood-plains buffer increases in flow. The flow spreads out over floodplain & energy absorbed; also healthy riparian vegetative cover in the watershed increase infiltration into soils and uptake of water (e.g., by plants) before it reaches the stream	Measure the **rate** of infiltration of water into soils OR discharge in stream in response to rain events (discharge = **rate** of water flow measured in volume per time…m^3/sec)	Without the benefits of floodplains, healthy stream corridor and watershed vegetation you see increased flood frequency and flood magnitude
Biodiversity support	All of the processes above contribute to the maintenance of biodiversity. For example, primary production and the flux of organic materials into streams help support diverse living assemblages	Measure the number of species and how abundance varies among them; this function is not a rate per se but because it is critical to the support of all other functions, it is included in the table.	Headwater streams support extremely high biodiversity and many rare species that contribute food for higher trophic levels and help maintain functions such as organic matter processing

2. Channel creation approach outside the scope of accepted science. The Chuitna mining project will destroy fully healthy Alaskan streams in a relatively pristine region as they clear approximately 5000 acres and mine through the streams. During this process they will divert water from the streams then dig through the streambed and surrounding area to remove more than 300 feet of 'overburden' so they can reach the coal seams. Many years later, they will replace the overburden, add topsoil back, and use heavy machinery to construct a channel that has similar dimensions (width, depth, slope, sinuosity, etc) to the one they destroyed. Thus they will attempt to create a stream after all the natural flow paths and landscape topography have been destroyed. This is not even in the realm of anything that has been scientifically tested and is certainly not within the realm of what is considered ecological restoration. In practice, ecological stream restoration varies along a continuum from: removing on-going impacts to a stream (e.g., preventing toxic inputs) and letting the system recover on its own; to enhancing in-stream habitat or the surrounding riparian zone (e.g., adding coarse woody debris to streams and planting vegetation) in an otherwise healthy stream; to full scale restoration that involves manipulations of an *existing* stream channel (e.g., re-grading banks and planting trees along a stream with eroding banks) (Williams et al. 1997, FISRWG 1998, Kaur and Chu 1999).

While the plan refers to this proposed mitigation project as "reconstruction", it is in fact a new channel construction or creation plan. While the latitude and longitude of the streams may be similar to what they were before, everything else that defines an ecologically healthy stream will be gone or will have been dramatically altered at the end of the mining period (e.g., flow paths, riparian soil and streambed biogeochemistry, groundwater-surface water (hyporheic) exchange rates, mature riparian vegetation, etc). The mining project is a major earth-moving project that will impact a significant amount of land. While they outline plans to attempt to restore recharge capacity of the lands, these plans are based on assumptions and model output that even if correct, in no way assure restoration of the hydrologic and biogeochemical properties that the riparian, streambed, surface water, and hyporheic zone flora and fauna presently rely on. There is no evidence provided that the groundwater-surface water exchange, the concentration of suspended sediments, or the water quality in the new channel will be similar to what is in the undisturbed streams presently.

[redacted]

Interestingly, from the plan prepared by PacRim, there seems to be a clear understanding that what they propose is outside the accepted range of current science and practice. From page 14 of the Fish Protection Plan:

> "Historically the majority of stream restoration practice has involved alterations of modifications of an existing water body that has been impaired in some way. Stream 2003 and other Chuitna Coal Mine area stream represent a more comprehensive restoration effort that includes 3-D restoration of the entire channel including floodplain structure and form" (page 14).

In other words, they acknowledge that what they propose is not typical NCD type restoration, which involves alterations of existing channels, but instead goes well beyond what has been done in other settings. This leaves us with determining if they provide any new evidence of the feasibility of their mitigation streams. They do not. There are no data provided in the plan, nor are there peer-reviewed scientific studies referenced that demonstrate healthy streams can be created after this level of impact to the land has occurred. Even with far less damage to a site, stream restoration projects that involve channel modification have an extremely high failure rate (Smith and Prestegaard 2005; Tullos et al. 2009; Palmer et al. 2009).

3. Morphologically based channel designs are not ecologically based. The mitigation plan argues that they will use the Natural Channel Design (NCD or Rosgen approach) approach for constructing stream 2003 which is based on 'tested methods' (page 29, Fish Protection Plan) that will "ensure pre-mined ecological functions and values are restored" (page 13). But the NCD

approach to stream restoration has in fact never been evaluated for its ecological effectiveness. This approach was designed by Rosgen (1994) to address channel stability based only on building a channel structure (shape, slope, etc.) that is able to transport the sediment and water inputs that are expected to be delivered to the stream prior to completion. There is no scientific evidence supporting the assumption that restoration of channel form will lead to full restoration of function (Palmer et al. 1997; Hilderbrand et al. 2005; Falk et al. 2006). How a stream looks (its form) is simply not the same as how it processes (its function: material and supports life, primary producers, invertebrates, etc.). The present mitigation plan provides no evidence that restoration of channel form will lead to restoration of function.

Not only does the plan assume that selection of a channel type (e.g., "D channel") from a channel classification scheme such as those proposed by Rosgen (1994) will necessarily result in full ecological restoration, but it also assumes that use of the NCD or Rosgen approach guarantees successful creation of a channel from a geomorphic and hydrologic perspective. However, channel designs based on a classification system that has not been fully evaluated at the site can lead to serious failures (Smith and Prestegaard 2005). As indicated in Palmer et al. (2005): "Attempts to develop restoration designs based on application of a single classification system across many environments have led to many failures in North America (e.g., Kondolf et al. 2001), because the specific processes and history of the river under study were not adequately understood." If the mitigation projects fail and channels are unstable, this could cause new environmental degradation. However, even if they are geomorphically stable, this does not address restoration of function. Indeed, the Rosgen scheme of classification does not deal with ecological functions at all.

The Rosgen classification is based on channel morphology and uses a hierarchical key to demarcate stream types based on specified ranges of quantitative variables, including entrenchment ratio, bankfull width:depth ratio, channel sinuosity, gradient, and dominant substrate (Rosgen 1994). While use of the Rosgen scheme for stream restoration has been very common in the past, current science (published in many peer-reviewed scientific journals) has documented numerous reasons that use of this scheme for restoration can be extremely problematic (Gillilan 1996; Miller and Ritter 1996; Shields et al. 1999; Doyle and Harbor 2000; Kondolf et al. 2001; Juracek and Fitzpatrick 2003; Niezgoda and Johnson 2005; Smith and Prestegaard 2005; Slate et al. 2007; Simon et al.

2007; Roper et al. 2008). In fact, an analysis of > 75 channel reconfiguration projects overwhelmingly showed that restoration of biodiversity failed (Palmer et al. 2009).

The fundamental problem with classification based restoration approaches is that they assume fixed endpoints and rigid classification schemes in which the type of stream desired can be achieved by constructing a specific channel form. Yet, streams are living systems – far more than rock-lined ditches. Even from a practical point of view, restoration is far more than creating some design based on external appearance. The fundamental distinction between form and function of stream channels is not acknowledged by the plan, which focuses on structural aspects of the channels and ignores functional aspects. The method in no way takes into account a whole array of biophysical factors that determine the ability of the channel to support all of the living resources in pristine streams in the area. Such factors include: intensity and duration of sunlight reaching the stream, which is determined in part by the vegetative structure; inputs of organic matter upon which the food web depends; nitrogen and carbon levels in the soil and streambed; etc.

4. Downstream impacts are not addressed. The mining activities will fundamentally and permanently alter the chemical, hydrologic and sediment regimes which are master variables controlling the suitability of running-water systems for supporting downstream reaches. Further, since watersheds act as a unit and a considerable amount of land in the watershed is to be cleared, the impacts are expected to extend far beyond the mined-through streams. Even if the overburden and topsoils are stored during the mining for use later during reclamation, they will not have the same biogeochemical properties as prior to disturbance. Many of the soils in this region are highly organic. Disturbing them will result in dramatic changes including different microbial communities, alterations to the soil C:N:P content and changes in porosity. Even given the planting (reclamation) that is proposed, there is no clear evidence that conditions would be likely to return to pre-mining mature conditions. Biogeochemical processes in watershed and riparian soils influence the delivery of nutrients and other materials to streams and because these processes are greatly influenced by the flow rates and paths that water takes as it moves through surface and subsurface soils, disruption of the magnitude proposed for this project will certainly alter water quality. Even actions that are less disruptive than the mining that is proposed such as road building and land clearing) are well known to influence water quality and the movement of materials to downstream waters (Naiman et al. 2005; Nadeau and Rains 2007).

Since the streams to be destroyed are primarily headwater streams which play disproportionate roles in nutrient processing and supporting biodiversity, it is particularly problematic that > 17 km of them will be destroyed and yet the ability to re-create them is unlikely. Because headwater streams provide food (organic matter and prey) for biota in downstream waters, when they are lost food supplies are likely to be inadequate downstream (Wipfli et al. 2007). This is particularly the case if surrounding riparian vegetation is lost, which will be the case for Creek 2003 and some of the other streams within the mining area. This could result in decreased growth of fish and other organisms (Freeman et al. 2007) in the Chuitna waters. Biota may also be subjected to higher rates of suspended sediment and other forms of water quality impairment since sediment flux and dissolved constituents post mining are unknown.

Thus, the massive disturbance to the entire watershed, but particularly headwater streams, will have impacts in downstream waters. Changes in key ecological processes can have cascading effects on nearby ecosystems, particularly in river networks where the movement of materials can effectively link very distant ecosystems (e.g. headwater streams to rivers). For this reason, ecologists have increasingly focused research on broader spatial contexts and multi-scale processes. Palmer 2009, Nakano and Murakami 2001 showed that across-habitat prey flux accounted for 25% and 44% of the total annual energy budget of bird and fish assemblages in a Japanese stream. Subsidies such as these have been extensively studied both theoretically and empirically (e.g. Polis and Hurd 1996).

Summary of Major Conclusions

This is a very large project that will seriously impact a sensitive ecosystem that currently supports highly valued fisheries, rare plant species, and ecologically important habitats including wetlands and streams. The wetland and stream ecosystems that will be lost or damaged perform essential functions in the provision of important ecosystem services and it is very unlikely (and no evidence is provided to indicate) that the ecosystem functions can be recovered. The surrounding habitats such as the tundra are extremely fragile and as the project proceeds the risk that surrounding habitats will suffer direct or indirect degradation will increase. Across all habitats, but particularly the aquatic

ones, biodiversity loss will be substantial yet this diversity is critical to ecological functions that ensure the long term health and stability of the ecosystems. Basic concerns about the mitigation plan fall into four categories. First, the applicants have not directly measured ecosystem functions and thus have not applied current science to the mitigation issues. Without these functional assessments, they do not know exactly what natural resource values are being lost and thus what they need to mitigate for. Second, the approach proposed for replacing the lost streams (especially Stream 2003) is outside the realm of stream restoration or rehabilitation practices. Their approach basically amounts to channel "creation" in an area in which the earth has been disturbed to depths of 300- 500 feet, the natural flow paths destroyed, and landscape topography reshaped. Channels that are created on these mined sites may have the same latitude and longitude of the original streams however everything else that defines an ecologically healthy stream will be gone or will have been dramatically altered. Third, the channel creation plans are based on a method (Natural Channel Design) that is morphologically based (reference templates) but there is no evidence that creation of channel form (pattern, plan, profile) will lead to restoration of ecological function. Indeed, there is ample evidence in the peer-reviewed literature that the approach they propose is problematic and that ecological outcomes typically fail. Fourth, impacts to the watershed and the headwater streams from the mining activities will fundamentally alter the chemical, hydrologic and sediment regimes which are master variables controlling the suitability of running-water systems for supporting downstream reaches.

In sum, based on the most current and rigorous science, the impacts of this project are very significant and there is no evidence that the restoration and mitigation plans that are proposed will either succeed or compensate for the natural resource losses.

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EXHIBIT 13

Report on Chuitna Coal Project of PacRim Coal

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Professor and Director
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Impacts from mining coal from the wetlands and forests above the Chuitna River will cause destruction of over 4,000 acres of wetlands and is highly likely to permanently change the ecosystem of the area and the productivity of the Chuitna River.

The ecosystem is a woven fabric of wetlands, tundra, forests, and tiny headwater streams that gather to build larger streams, to eventually pour into the Chuitna River. Forty-one percent of the watershed will be directly impacted from mining and backfilling of the mine. What occurs in these headwaters, wetlands, tundra, and forests is vital to the water quality and the fish downstream. It is in these areas that carbon is stored and nutrients are cycled from detritus to microbes, from microbes to insects. The wetlands in particular are vital to storing water that seeps down into flow paths beneath the earth, to surface at the bottom of streams, keeping them flowing when there is no rain or snow. As water trickles through wetlands, microbes in the muck and peat remove heavy metals and purify the water. Wetlands are the source of both pure water and primary nutrients such as carbon, nitrogen, and phosphorous which make up the very base of the food chain.

Wetlands water seeps up to become headwater streams, disproportionately rich in biodiversity for their small size, and the source of much of the food that arrives downstream. Headwaters provide breeding and nursery grounds for insects that spend the rest of their lives in larger streams and rivers, and are an important food source for fish. Headwaters provide spawning grounds and help to regulate stream temperature. The rich biodiversity found here buffers the streams so they recover more rapidly from rapid changes such as climate swings, flooding, and human damage.

Tundra, wetlands, and headwater streams will all be destroyed during mining. And there is little chance they will be restored. Tundra is very sensitive and only revives when specific conditions are met, including maintaining corridors to more tundra throughout the mining process. Wetlands and headwaters cannot be restored to ecological function if the very material that they rely on -- deep sediment structure and long-entrained flow paths -- are mined through, ground up, and replaced in the mining pit as a relatively homogenous pile of rubble and dirt.

One stream, "Stream 2003" also called Middle Creek, will be completely destroyed. It will not be "impacted", but rather mining will go down hundreds of feet beneath it, completely removing the stream bed and any remnant of the stream for 11 miles. While stream reconstruction has been done successfully by re-grading and re-vegetating banks, or adding or removing debris to create habitat, no one has simply created a new stream where none exists. A new ditch can be dug where the old stream used to be, and can have the same curves and shape. But it will not

have the exchange of surface and groundwater at the streambed, upwelling areas for fish to lay their eggs in, biodiversity of insects that headwater streams provide as food for fish, the purity of water and nutrients wetlands provided.

Nor is PacRim attempting to assess the functions of the stream and its associated ecosystems as they are now. Without such an assessment – rates of nutrient cycling, flood control, sediment control, water purification, and more – PacRim has no end goal to attempt to reach.

In summary, there are three main areas of concern with the mitigation plan:

First, the applicants have not directly measured ecosystem functions and thus have not applied current science to the mitigation issues. Without these functional assessments, they do not know exactly what natural resource values are being lost and thus what they need to mitigate for. Second, the approach proposed for replacing the lost streams (especially Stream 2003) is outside the realm of stream restoration or rehabilitation practices. Their approach basically amounts to channel "creation" in an area in which the earth has been disturbed to depths of 300- 500 feet, the natural flow paths destroyed, and landscape topography reshaped. Indeed, there is ample evidence in the peer-reviewed literature that the approach they propose (Natural Channel Design) typically fail ecologically. Third, impacts to the watershed and the headwater streams from the mining activities will fundamentally alter the chemical, hydrologic and sediment regimes which are master variables controlling the water quality and productivity downstream.

In sum, based on the most current and rigorous science, the impacts of this project are very significant and there is no evidence that the restoration and mitigation plans that are proposed will compensate for the natural resource losses.

EXHIBIT 14

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

Effects of Coal-Mine Drainage on Stream Water Quality in the Allegheny and Monongahela River Basins—Sulfate Transport and Trends

by James . Sams III and Kevin M. Beer

Water-Resources Investigations Report 99-4018

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

Lemoyne, Pennsylvania
2000

U.S. DEPARTMENT OF THE INTERIOR

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FOREWORD

The mission of the U.S. Geological Survey (USGS) is to assess the quantity and quality of the earth resources of the Nation and to provide information that will assist resource managers and policymakers at Federal, State, and local levels in making sound decisions. Assessment of water-quality conditions and trends is an important part of this overall mission.

One of the greatest challenges faced by water-resources scientists is acquiring reliable information that will guide the use and protection of the Nation's water resources. That challenge is being addressed by Federal, State, interstate, and local water-resource agencies and by many academic institutions. These organizations are collecting water-quality data for a host of purposes that include: compliance with permits and water-supply standards; development of remediation plans for a specific contamination problem; operational decisions on industrial, wastewater, or water-supply facilities; and research on factors that affect water quality. An additional need for water-quality information is to provide a basis on which regional and national-level policy decisions can be based. Wise decisions must be based on sound information. As a society we need to know whether certain types of water-quality problems are isolated or ubiquitous, whether there are significant differences in conditions among regions, whether the conditions are changing over time, and why these conditions change from place to place and over time. The information can be used to help determine the efficacy of existing water-quality policies and to help analysts determine the need for, and likely consequences of, new policies.

To address these needs, the Congress appropriated funds in 1986 for the USGS to begin a pilot program in seven project areas to develop and refine the National Water-Quality Assessment (NAWQA) Program. In 1991, the USGS began full implementation of the program. The NAWQA Program builds upon an existing base of water-quality studies of the USGS, as well as those of other Federal, State, and local agencies. The objectives of the NAWQA Program are to:

- Describe current water-quality conditions for a large part of the Nation's freshwater streams, rivers, and aquifers.
- Describe how water quality is changing over time.
- Improve understanding of the primary natural and human factors that affect water-quality conditions.

This information will help support the development and evaluation of management, regulatory, and monitoring decisions by other Federal, State, and local agencies to protect, use, and enhance water resources.

The goals of the NAWQA Program are being achieved through ongoing and proposed investigations of 60 of the Nation's most important river basins and aquifer systems, which are referred to as study units. These study units are distributed throughout the Nation and cover a diversity of hydrogeologic settings. More than two-thirds of the Nation's freshwater use occurs within the 60 study units and more than two-thirds of the people served by public water-supply systems live within their boundaries.

National synthesis of data analysis, based on aggregation of comparable information obtained from the study units, is a major component of the program. This effort focuses on selected water quality topics using nationally consistent information. Comparative studies will explain differences and similarities in observed water-quality conditions among study areas and will identify changes and trends and their causes. The first topics addressed by the national synthesis are pesticides, nutrients, volatile organic compounds, and aquatic biology. Discussions on these and other water-quality topics will be published in periodic summaries of the quality of the Nation's ground and surface water as the information becomes available.

This report is an element of the comprehensive body of information developed as part of the NAWQA Program. The program depends heavily on the advice, cooperation, and information from many Federal, State, interstate, Tribal, and local agencies and the public. The assistance and suggestions of all are greatly appreciated.

Robert M. Hirsch
Chief Hydrologist

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CONVERSION FACTORS AND ABBREVIATIONS

Multiply	By	To obtain
	Length	
inch (in)	2.54	centimeter
mile (mi)	1.609	kilometer
	Area	
square mile (mi^2)	2.590	square kilometer
	Mass	
ton, short (2,000 lb)	0.9072	megagram
ton per day (ton/d)	0.9072	metric ton per day
ton per square mile (ton/mi^2)	0.3503	megagram per square kilometer

Abbreviations water-quality units used in this report:

mg/L, milligrams per liter

$ton/mi^2/yr$ - ton per square mile per year

EFFECTS OF COAL-MINE DRAINAGE ON STREAM WATER QUALITY IN THE ALLEGHENY AND MONONGAHELA RIVER BASINS—SULFATE TRANSPORT AND TRENDS

by James I. Sams III and Kevin M. Beer

ABSTRACT

In 1980, the Allegheny and Monongahela Rivers transported a sulfate load of 1.2 million and 1.35 million tons, respectively, to the Ohio River at Pittsburgh. The Monongahela River Basin had a sulfate yield of 184 tons per square mile per year compared to 105 tons per square mile per year for the Allegheny River Basin. Within the large Allegheny and Monongahela River Basins, the subbasins with the highest sulfate yields in tons per square mile per year were those of Redstone Creek (580), Blacklick Creek (524), Conemaugh River (292), Buffalo Creek (247), Stonycreek River (239), Two Lick Creek (231), Dunkard Creek (212), and Loyalhanna Creek (196). These basins have been extensively mined. The sulfate yields of Brokenstraw and Conewango Creeks, which are outside the area underlain by coal and thus contain no coal mines, were 25 and 24 tons per square mile per year, respectively.

Within the Allegheny and Monongahela River Basins, seven sites showed significant trends in sulfate concentration from 1965 to 1995. Dunkard Creek and Stonycreek River show significant upward trends in sulfate concentration. These trends appear to be related to increases in coal production in the two basins from 1965 to 1995. Blacklick Creek at Josephine and Loyalhanna Creek at Loyalhanna Dam show significant downward trends in sulfate concentration between 1965 and 1995. Blacklick Creek had a 50-percent decrease in sulfate concentration. Coal production in the Blacklick Creek Basin, which reached its peak at almost 4 million tons per year in the 1940's, dropped to less than 1 million tons per year by 1995. In the Loyalhanna Creek Basin, which had a 41-percent decrease in sulfate concentration, coal-production rates dropped steadily from more than 1.5 million tons per year in the 1940's to less than 200,000 tons per year in 1995.

INTRODUCTION

The Allegheny and Monongahela River Basins collectively form 1 of 59 study units selected nationwide as part of the U.S. Geological Survey's (USGS) National Water Quality Assessment (NAWQA) Program. The NAWQA Program was developed to provide nationally consistent information on the status and trends of the Nation's water quality in an effort to evaluate past water-quality problems and to provide a base of knowledge for making future water-management decisions. Studies of the hydrologic systems that include parts of most major river basins and aquifer systems (study-unit investigations) are the building blocks of the NAWQA Program. Study units range in size from less than 1,000 to more than 60,000 mi^2. The NAWQA Program uses a multiscale, integrated, and interdisciplinary approach to water-quality assessment. Studies are designed to examine the physical, chemical, and biological aspects of each basin. Results from these investigations enable the USGS to meet its program goals by describing current water-quality conditions and identifying natural and human factors affecting basin water quality (Hirsch and others, 1988).

One aspect of the NAWQA Program is to analyze existing water-quality conditions throughout the study-unit. Acid mine drainage (AMD) from coal mining has been identified as the factor having the most widespread effect on water quality in the

Allegheny and Monongahela River Basins (U.S. Army Corps of Engineers, 1997). Most of the drainage is from abandoned mine sites. Water discharging from deep-mine openings and surface-mine seeps results in elevated concentrations of acidity, iron, manganese, aluminum, and sulfate in receiving streams and rivers. Streams and rivers with low buffer capacity are not able to neutralize the acid load and consequently become acidic. An estimated 2,390 mi of streams in the Allegheny and Monongahela River Basins have been degraded by AMD to the point of not being able to support fish communities (U.S. Environmental Protection Agency, 1995). The AMD problem is a consequence of more than 200 years of coal mining in the study unit. On the basis of current production rates, available coal reserves, and mining economics, coal mining should continue in the Allegheny and Monongahela River Basins well into the next century. For more than two decades, legislation and programs have been in place at the national and state levels to monitor and reduce the effects of coal mining on water quality.

Purpose and Scope

This report examines sulfate transport and trends as an indicator of drainage from past and present coal mining in the Allegheny and Monongahela River Basins. This report also presents a brief history of coal-mining activities in the study area and a description of the environmental effects associated with mine drainage. A description of the study area and analytical methods used in this investigation are included. To evaluate the spatial distribution of mining effects in the Allegheny and Monongahela River Basins, annual sulfate loads and yields were calculated for 37 water quality monitoring stations for the 1980 water year. Trends in sulfate loads are presented for 15 of the 37 sites for the period 1965 to 1995. This report will be useful to managers of water resources for evaluating the effects of mining on water quality and evaluating the effectiveness of AMD control measures.

Description of Study Area

The Allegheny and Monongahela River Basins are mostly within the Appalachian Plateaus Physiographic Province within the states of New York, Pennsylvania, Maryland, and West Virginia (fig. 1). The total combined drainage area of the Allegheny and Monongahela River Basins is 19,145 mi^2. The Allegheny River has a length of 295 mi; the Monongahela River has a length of 116 mi. The Allegheny and Monongahela Rivers join to form the Ohio River at an area in Pittsburgh known as the Point.

The climate in the study area is humid continental, characterized by warm summers and cold winters. Prevailing winds are from the west. Average annual precipitation over the study area is 42 in. and ranges from 37 in. in the northern areas to 60 in. in the southern mountain areas. Average annual runoff is 18 to 26 in. in the northern part of the study area and 25 to 40 in. in the southern mountains.

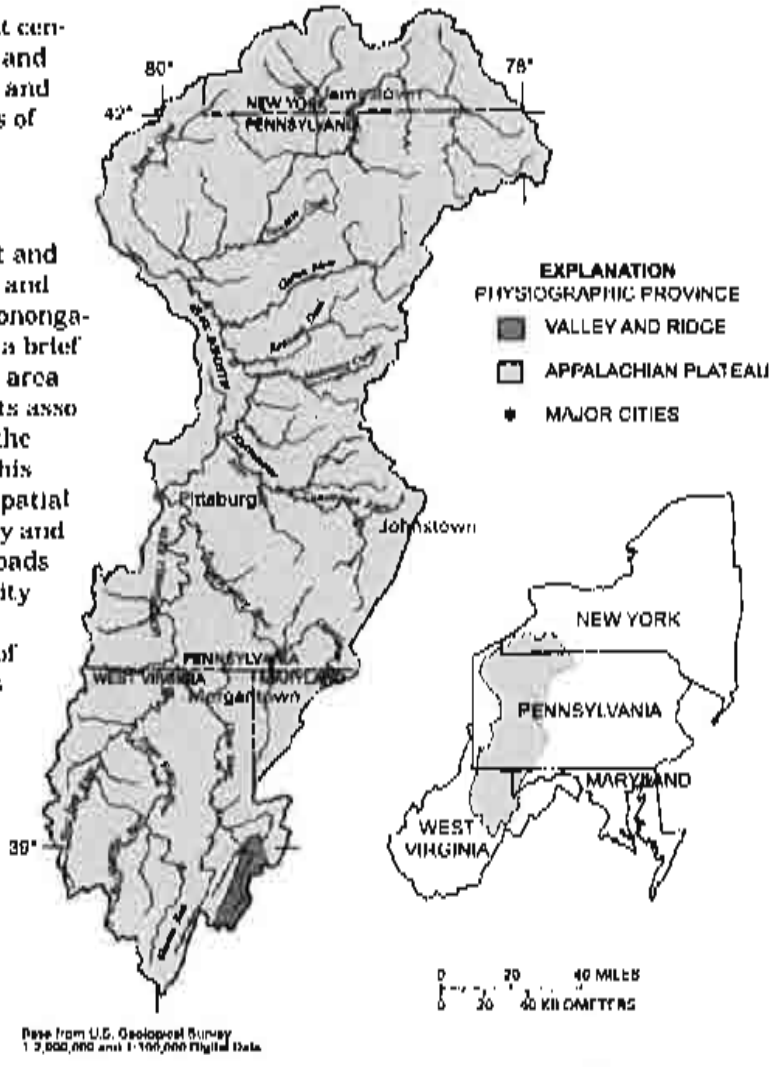

Figure 1. Location of the study area, major cities, and streams in the Allegheny and Monongahela River Basins.

The study area is a mixed land-use type containing about 74 percent forest, 21 percent mixed agriculture, and 2 percent urban. The remaining area consists of open water, wetlands, and barren land (from mining, construction, etc.).

The study area is underlain by sedimentary rocks of Devonian, Mississippian, Pennsylvanian, and Permian age rocks. These rocks are commonly fractured and folded but may also be flat-lying. Bituminous coal is present in the Mississippian, Pennsylvanian, and Permian rocks and underlies approximately 70 percent of the study area.

ACKNOWLEDGMENTS

The authors wish to thank Michael Langland of the USGS in Lemoyne, Pa., for his assistance in setting up and applying the computer model used to estimate sulfate loads and trends.

COAL MINING IN THE ALLEGHENY AND MONONGAHELA BASINS

History of Mining Activities

Coal has been mined from the Allegheny and Monongahela River Basins for more than 200 years. Early records of coal mining in Pennsylvania are shown on plans of Fort Pitt in 1761. Fort Pitt was located at the confluence of the Allegheny and Monongahela Rivers at present day Pittsburgh. The most significant coal bed in the basin is the Pittsburgh seam, which also is one of the most valuable mineable deposits in the world for its use in the iron and steel industry (Edmunds and Koppe, 1968).

Coal has been central to the economy and lifestyle of many communities in the Allegheny and Monongahela River Basins. More than any other activity, coal mining has influenced growth and development of this region. Beginning in the 1800's, the mining of coal helped fuel an industrial revolution that would make Pittsburgh one of the wealthiest cities in the United States. The mining of coal in western Pennsylvania and the iron and steel manufacturing in Pittsburgh provided thousands of jobs for immigrants to this country. Coal production in Pennsylvania peaked at 177 million tons in 1918, at the height of World War I. In 1920, coal supplied 80 percent of the total United States fuel needs (Western Pennsylvania Conservancy, 1980). The Great Depression of the 1930's began a trend of declining coal production in Pennsylvania. Competing energy sources included fuel oil, natural gas, and nuclear power. In 1974, there was a resurgence in coal mining in Pennsylvania. The increase in production was short lived, however, due in part to the passage of strict environmental laws such as the Federal Surface Mining Control and Reclamation Act of 1977 (P.L. 95-87) (SMCRA) (Pennsylvania Coal Association, 1994). In 1995, Pennsylvania was the fourth largest coal producer in the United States at 69 million tons, behind Wyoming, West Virginia, and Kentucky. Until the late 1940's, most coal had been mined underground using room and pillar methods. But modern earthmoving equipment developed in the late 1940's resulted in an increase in coal mined using surface-mining methods. Drake (1931) reported that streams in the coal regions were being seriously degraded by mine drainage to the point of being unusable for domestic and industrial water supplies.

Environmental Consequences of Mine Drainage

Formation of Acid Mine Drainage

Acid mine drainage (AMD) results from the formation of sulfuric acid in the oxidation of iron-sulfide minerals such as pyrite. Pyrite can be found in the coal seam and shale and sandstone strata adjacent to the coal beds. During coal mining, pyrite exposed to air and water oxidizes to form ferrous sulfate and sulfuric acid. The reaction can proceed to produce ferric sulfate, ferric hydroxide, and more sulfuric acid. The following equations show the generally accepted sequence of pyrite reactions (U.S. Office of Surface Mining, 1998a):

$$2\,FeS_2 + 7\,O_2 + 2\,H_2O \rightarrow 2\,Fe^{2+} + 4\,SO_4^{2-} + 4\,H^+ \quad (1)$$

$$4\,Fe^{2+} + O_2 + 4\,H^+ \rightarrow 4\,Fe^{3+} + 2\,H_2O \quad (2)$$

$$4\,Fe^{3+} + 12\,H_2O \rightarrow 4\,Fe(OH)_3 + 12\,H^+ \quad (3)$$

$$FeS_2 + 14\,Fe^{3+} + 8\,H_2O \rightarrow 15\,Fe^{2+} + 2\,SO_4^{2-} + 16\,H^+ \quad (4)$$

The AMD chemical reactions produce elevated concentrations of the insoluble precipitate ferric hydroxide [$Fe(OH)_3$], dissolved sulfate (SO_4^{2-}), and acid (H^+). The pyrite weathering process is a series of chemical reactions but also has an important microbiological component. This reaction can be greatly accelerated by a species of bacteria, *Thiobacillus ferrooxidans* (Singer and Stumm, 1970). Secondary reaction of the sulfuric acid with compounds in adjacent rocks or mine spoil can produce high

concentrations of aluminum, manganese, zinc, and other constituents in mine drainage waters (Tolar, 1982). As pyrite is oxidized, oxidation products are transported from the oxidation site through subsurface flow systems by ground water or infiltrating precipitation to receiving streams. Acidic water produced by the AMD chemical reactions may persist for only a short time if sufficient alkalinity is available to neutralize the acid. Once the neutralization capacity is exceeded, however, acid begins to accumulate and the pH decreases.

The watershed represents an integration of many AMD processes throughout the basin. The AMD process consists of three phases: initiation, propagation, and termination. The initiation phase can begin as soon as pyritic materials are exposed to an oxidizing environment; however, the acid load generated is relatively small. In the propagation phase, acid production increases rapidly. In the termination phase, acid production gradually declines. The actual times associated with these phases are uncertain but generally are on the order of years to decades. Modeling predictions and comparison to a limited number of field sites indicate that the acid production peaks 5 to 10 years after mining, followed by a gradual decline over 20 to 40 years (Ziemkiewicz and Meek, 1994).

This cyclic propagation of acid generation takes place very rapidly and continues until the supply of exposed sulfuric material is exhausted. Although natural weathering reduces sulfuric material available for AMD reactions, the opening of new mines and roof falls of old mines create new surfaces for AMD reactions. Other pyrite sources include waste material piled outside of mine openings, refuse piles from preparation plants, and surface-mine overburden that has been backfilled after coal removal.

The geologic setting of the mine site is a major factor in post-mining water chemistry. Post-mining water quality can range from alkaline to severely acidic depending on the relative amounts of carbonates and sulfides. Carbonates weather to produce alkalinity. The weathering of sulfides such as pyrite produces acidity. The distribution of carbonates and sulfides throughout the coal regions in this study area is related to the depositional environments in which sediments accumulated. High concentrations of sulfide minerals occur in marine and brackish environments, whereas freshwater deposits frequently contain calcareous minerals. In the coal bearing rocks of western Pennsylvania, the amount of calcareous rocks increase stratigraphically upward from the Pottsville Group (less than 1 percent calcareous rock) to rocks of the Conemaugh group, which can be greater than 50 percent calcareous (Pennsylvania Department of Environmental Protection, 1998). If sufficient carbonate is present, post mining water quality will be alkaline.

An understanding of mine-site geology, rock composition, and depth of weathering is important in estimating post-mining water quality. Changes to the lateral and vertical facies of rock units occur within mine sites and thus require detailed drilling to map these changes. At the watershed scale, many geologic factors influence water quality. It was beyond the scope of this investigation to collect and evaluate geologic data.

Since 1979, mining has been regulated by the SMCRA, which has many provisions designed to limit environmental effects from coal mining. The law requires coal operators and state regulatory authorities to evaluate probable hydrologic consequences of the mining operation. Methods have been developed to estimate post mining water quality through chemical analysis of overburden such as acid-base accounting methods (Sobek and others, 1978). This technique is used to identify alkaline and acidic strata. The weight percent of sulfur in stratigraphic layers is measured to estimate the amount of pyritic sulfur and the maximum potential acidity. Since the mid 1980's, practices have been developed to handle toxic overburden layers in a special manner to reduce AMD chemical reactions. These advances in mining technology have resulted in a greater percentage of completed coal mines with acceptable post-mining water quality. For example, the Pennsylvania Department of Mining and Reclamation had an average failure rate of 14.7 percent (mine treatment required) for permits issued from 1977 to 1984 compared to a failure rate of 2.9 percent for permits issued from 1985 to 1992 (Pennsylvania Department of Environmental Protection, 1998).

Effects of Mine Drainage on Streams and Rivers

A study by Williams and others (1996) on 270 mine discharges in the Stonycreek River Basin found that many of the discharges had a pH less than 3.0. Water samples from these discharges generally had high concentrations of acidity, iron, manganese, aluminum, and sulfate. Water quality is thus severely degraded when mine discharges such

as these enter streams and rivers. A common scene throughout the coal regions of Pennsylvania and West Virginia is shown in figure 2.

AMD can seriously affect aquatic habitats. Stream bottoms covered with orange or yellow-brown iron oxide or white aluminum oxide precipitates can be toxic to benthic algae, invertebrates, and fish. Bottom-dwelling organisms are particularly sensitive to AMD precipitates. Depressed food supplies, gill clogging, and general smothering by iron or aluminum precipitates, along with direct toxicity from ingested metals, contribute to the decline of benthic invertebrates in metal oxide polluted streams (Koryak, 1997).

The effects of AMD on streams have been documented for many years. In 1906, a large fish kill on the Allegheny River below the Kiskiminetas River confluence was caused by AMD (Lewis, 1906). In the early 1930's, the U.S. Public Health Service designed a program to seal abandoned coal mines in an attempt to prevent AMD formation. Tisdale (1936) reported that 3,644 mine openings had been sealed in West Virginia, which improved water quality in 500 stream miles.

Figure 2. A stream affected by mine drainage in southwestern Pennsylvania.

The understanding of mine-site hydrology and the factors affecting water quality has greatly reduced post-mining water-quality problems for recently completed mine sites. However, mine drainage from abandoned mines remains a major problem in streams and rivers in the Allegheny and Monongahela River Basins. The cost to clean up the AMD problem from abandoned mine-land sites in Pennsylvania is estimated at 5 billion dollars, and the estimated cost to reclaim all abandoned mine-land sites is 15 billion dollars (Pennsylvania Department of Environmental Protection, 1996). Pennsylvania currently receives about 19 million dollars per year from the Federal Office of Surface Mining (OSM) to correct abandoned mine-land problems. Most of this money is spent to correct unsafe conditions such as open mine shafts (U.S. Office of Surface Mining, 1998b).

STUDY APPROACH AND METHODS

Database and Mining Indicator Parameter

Sulfate was selected as an indicator of the presence of AMD in streams in the Allegheny and Monongahela River Basins. Sulfate data from water-quality monitoring stations in Pennsylvania and West Virginia were analyzed to evaluate the long-term trends in the effects of coal mining on the streams. These data were retrieved from the USEPA Storage and Retrieval of U.S. Waterways Parametric Data (STORET) database. The monitoring stations were part of the Pennsylvania and West Virginia Water Quality Monitoring Network (WQN) Program. Sites were selected on the basis of the availability of continuous streamflow data. The streamflow data were retrieved from the USGS Water Data Storage and Retrieval System (WATSTORE) database. Sites used in this study are shown on figure 3 and listed in table 1. The 1980 water year was selected to look at the spatial distribution of mining effects because that year had the greatest number of WQN sites (37) with data between 1965 and 1995. Fifteen of the sites had sulfate and streamflow data from 1965 to 1995. These sites were used to evaluate long-term trends of mine drainage using sulfate as the indicator.

Sulfate is a conservative compound and therefore a good indicator of the occurrence and intensity of mine drainage. Concentrations of sulfate usually are not high in water draining unmined areas even where pyrite is abundant in the coal and associated strata. As an example, water samples collected from Stony Fork near Gibbon Glade, Pa., from 1978 to 1994 show relatively low concentrations of sulfate prior to 1985, when coal mining began in the basin (fig. 4). The median concentration of sulfate in samples collected after mining began was about five times greater than the median concentration in samples collected before mining. The range in streamflow conditions at the time water samples were collected was similar for pre- and post-mining periods. As described earlier, the mining process exposes pyrite to weathering, which can release acidity, iron, manganese, aluminum, and sulfate. Although mine drainage may contain elevated concentrations of iron, manganese, and aluminum, these elements usually are not a reliable indicator of AMD because they may not remain in solution. Sulfate, however, is an excellent indicator of mine drainage because the sulfate ion is very soluable and chemically stable at the pH levels normally encountered in natural waters (Hem, 1985). The treatment of mine drainage to remove metals and neutralize acidity has little or no effect on sulfate concentration (Keith Brady, Pennsylvania Department of Environmental Protection, oral commun., 1997).

Base from U.S. Geological Survey
1:7,000,000 and 1:100,000 Digital Data

Figure 3. Locations of water-quality monitoring sites.

LOAD ESTIMATIONS

Annual sulfate loads were computed from sulfate concentration data by use of a 7-parameter log-linear multiple regression model, known as Estimator, which was developed by Cohn and others (1989). The model was validated by Cohn and others (1992) with repeated split-sample studies and is currently used at nine monitoring stations for the Chesapeake Bay Nutrient Monitoring Program (Maryland Department of the Environment, 1992). The model is developed from measured streamflow and concentration data. Continuous daily mean streamflow data are used in the derived regression model to calculate mean daily chemical concentrations. The model is a multiple regression equation of the form:

$$\ln[C] = \beta_0 + \beta_1 \ln[Q/\bar{Q}] + \beta_2 \ln\{[Q/\bar{Q}]\}^2 + \beta_3[T - \bar{T}] + \beta_4[T - \bar{T}]^2 + \beta_5 \sin[2\pi T] + \beta_6 \cos[2\pi T] + \varepsilon \qquad (5)$$

Table 1. Water-quality sites for analysis of mine drainage effects in the Allegheny and Monongahela River Basins

Site	Drainage area (square miles)	U.S. Geological Survey site identification number	State water quality network identification number
Allegheny River Basin			
Potato Creek at Smethport, Pa.	160	03009680	WQN0856
Allegheny River at Eldred, Pa.	550	03010500	WQN0807
Kinzua Creek near Guffey, Pa.	46	03011800	WQN0855
Conewango Creek at Russell, Pa.	816	03015000	WQN0832
Brokenstraw Creek at Youngsville, Pa.	321	03015500	WQN0831
Allegheny River at West Hickory, Pa.	3,660	03016000	WQN0805
French Creek at Utica, Pa.	1,028	03024000	WQN0826
Allegheny River at Franklin, Pa.	5,982	03025500	WQN0804
West Branch Clarion River at Wilcox, Pa.	63	03028000	WQN0824
Clarion River at Cooksburg, Pa.	807	03029500	WQN0822
Clarion River near Piney, Pa.	951	03030500	WQN0821
Allegheny River at Parker, Pa.	7,671	03031500	WQN0803
Redbank Creek at St. Charles, Pa.	528	03032500	WQN0820
Allegheny River at Kittanning, Pa.	8,973	03036500	WQN0802
Stonycreek River at Ferndale, Pa.	451	03040000	WQN0817
Conemaugh River at Seward, Pa.	715	03041500	WQN0811
Blacklick Creek at Josephine, Pa.	192	03042000	WQN0814
Two Lick Creek at Graceton, Pa.	171	03042500	WQN0815
Loyalhanna Creek at Kingston, Pa.	172	03045000	WQN0813
Loyalhanna Creek at Loyalhanna Dam, Pa.	292	03047000	WQN0812
Buffalo Creek near Freeport, Pa.	137	03049000	WQN0808
Allegheny River at New Kensington, Pa.	11,410	03049500	03049625
Monongahela River Basin			
Buckhannon River at Hall, W. Va.	277	03053500	550796
Tygart Valley River at Philippi, W. Va.	916	03054500	550822
Tygart Valley River at Colfax, W. Va.	1,366	03057000	550574
Dry Fork at Hendricks, W. Va.	345	03065000	550851
Shavers Fork at Parsons, W. Va.	214	03069000	550486
Dunkard Creek at Shannopin, Pa.	229	03072000	WQN0714
Monongahela River at Greensboro, Pa.	4,407	03072500	WQN0703
South Fork Tenmile Creek at Jefferson, Pa.	180	03073000	WQN0713
Redstone Creek at Waltersburg, Pa.	74	03074500	WQN0712
Casselman River at Grantsville, Md.	63	03078000	CAS0479
Casselman River at Markleton, Pa.	382	03079000	WQN0710
Laurel Hill Creek at Ursina, Pa.	121	03080000	WQN0724
Youghiogheny River at Connellsville, Pa.	1,326	03082500	WQN0707
Youghiogheny River at Sutersville, Pa.	1,715	03083500	WQN0706
Monongahela River at Braddock, Pa.	7337	03085000	03085000

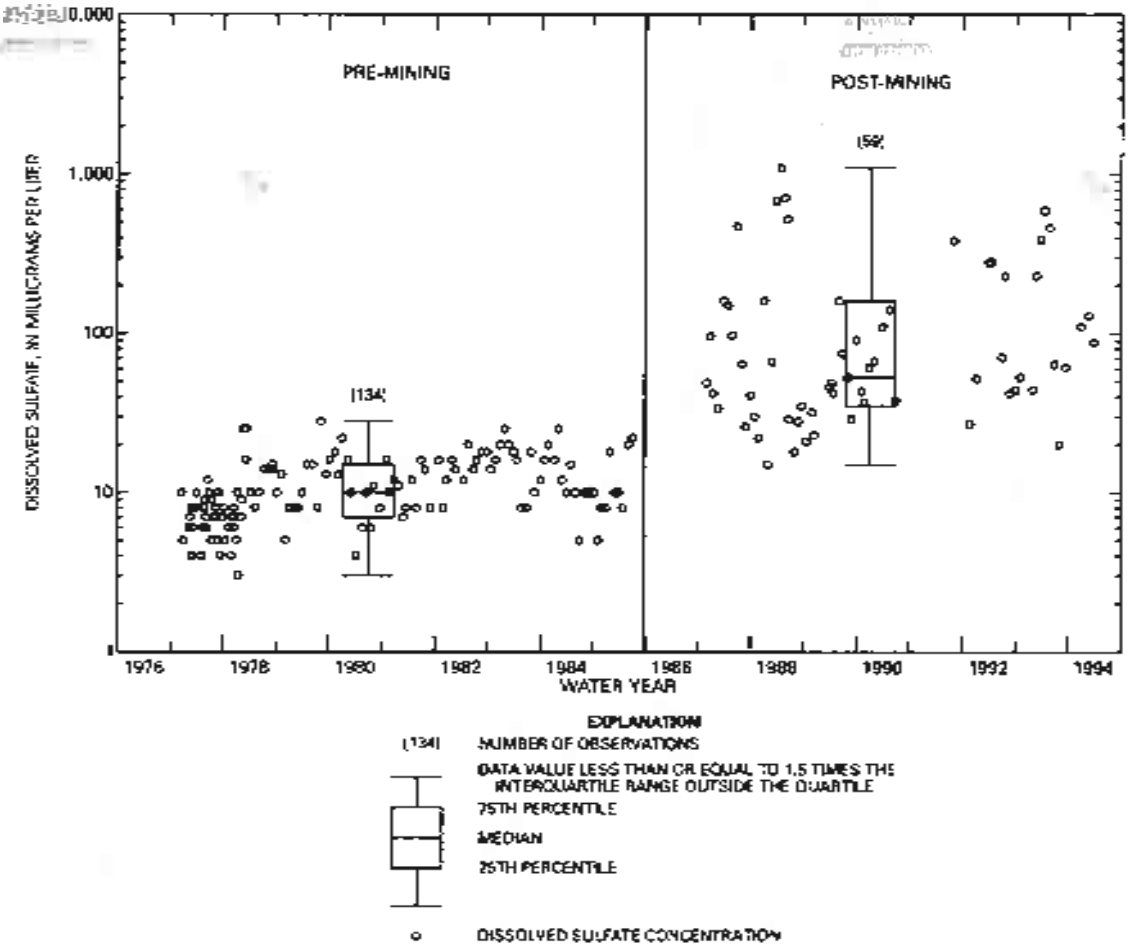

Figure 4. Concentrations of dissolved sulfate in pre-mining and post-mining periods at Stony Fork near Gibbon Glade, Pa. (1977-94).

Mean daily concentrations predicted by the model are combined with mean daily streamflow to calculate daily loads as follows:

$$L_T = \sum_{i=1}^{T} (C_{pi} \times Q_i \times K) \qquad (6)$$

where L_T is calculated load over time interval T for constituent L,

C_{pi} is predicted concentration of constituent i for day i in milligrams per liter (calculated by the model),

Q_i is measured mean daily streamflow for day i in cubic feet per second, and

K is conversion factor $2.699 \times 10^{-3} \frac{s \times L \times ton}{ft^3 \times mg \times d}$, where s is seconds, L is liters, ton is tons, ft^3 is cubic feet, mg is milligrams, and d is days.

(The model usually reports estimated loads in kilograms per day; for this study, the K listed above converts kilograms per day to tons per day.)

SULFATE TRANSPORT (SPATIAL DISTRIBUTION 1980)

During 1980, the Allegheny and Monongahela Rivers transported a sulfate load of 1.2 million and 1.35 million tons, respectively, to the Ohio River at Pittsburgh (table 2). The Monongahela River Basin, although smaller in drainage area (7,337 mi^2) compared to the Allegheny River Basin (11,410 mi^2), contributed 53 percent of the sulfate load to the Ohio River at Pittsburgh in 1980.

Fifty-five percent (661,800 tons) of the total sulfate load of the Allegheny River enters the river above the WQN site at Kittanning (table 2). The Allegheny River at Parker transported 270,200 tons and Redbank Creek at St. Charles transported 61,480 tons for a combined total of 331,680 tons, or 50 percent of the sulfate load at Kittanning. The remaining sulfate load below Parker and above Kittanning likely enters the Allegheny River from mine-drainage sources on Mahoning Creek, lower sections of Redbank Creek below St. Charles, or from small streams and tributaries between Parker and Kittanning. The Kiskiminetas River enters the Allegheny River about 15 mi below Kittanning. Sulfate transport within the Kiskiminetas River from

Table 2. *Sulfate loads, yields, and percentages for Water Quality Network Stations in the Allegheny and Monongahela River Basins (1980 water year)*

STATION NAME	SULFATE LOAD (TONS)	RANK	SULFATE YIELD (TONS PER SQUARE MILE)	RANK	PERCENTAGE OF TOTAL SULFATE LOAD AT OUTFLOW
Allegheny Basin					
Potato Creek at Smethport, Pa.	2,555	20	10	21	0.21
Allegheny River at Eldred, Pa.	8,961	18	16	20	.75
Kinzua Creek near Guffey, Pa.	794	22	17	19	.07
Conewango Creek at Russell, Pa.	19,420	16	24	16	1.62
Brokenstraw Creek at Youngsville, Pa.	8,050	19	25	15	.67
Allegheny River at West Hickory, Pa.	70,360	9	19	18	5.85
French Creek at Utica, Pa.	31,690	15	31	14	2.64
Allegheny River at Franklin, Pa.	115,400	5	19	17	9.60
West Branch Clarion River at Wilcox, Pa.	888	21	14	22	.07
Clarion River at Cooksburg, Pa.	39,400	13	49	12	3.28
Clarion River near Piney, Pa.	91,110	8	96	9	7.58
Allegheny River at Parker, Pa.	270,200	3	35	13	22.48
Redbank Creek at St. Charles, Pa.	61,480	10	116	7	5.11
Allegheny River at Kittanning, Pa.	661,800	2	74	10	55.06
Stonycreek River at Ferndale, Pa.	107,800	6	230	4	8.97
Conemaugh River at Seward, Pa.	208,600	4	292	2	17.35
Blacklick Creek at Josephine, Pa.	100,600	7	524	1	8.37
Two Lick Creek at Graceton, Pa.	39,540	12	231	5	3.29
Loyalhanna Creek at Kingston, Pa.	9,250	17	54	11	.77
Loyalhanna Creek at Loyalhanna Dam, Pa.	57,220	11	196	6	4.76
Buffalo Creek near Freeport, Pa.	33,845	14	247	3	2.82
Allegheny River at New Kensington, Pa.	1,202,000	1	105	8	100.00
Monongahela Basin					
Buckhannon River at Hall, W. Va.	21,300	11	77	8	1.58
Tygart Valley River at Philippi, W. Va.	61,630	6	67	10	4.57
Tygart Valley River at Colfax, W. Va.	93,120	4	68	9	6.90
Dry Fork at Hendricks, W. Va.	39,700	9	115	5	2.94
Shavers Fork at Parsons, W. Va.	6,462	13	30	15	.48
Dunkard Creek at Shannopin, Pa.	48,450	7	212	2	3.59
Monongahela River at Greensboro, Pa.	853,300	2	194	3	63.21
South Fork Tenmile Creek at Jefferson, Pa.	11,370	12	63	12	.84
Redstone Creek at Waltersburg, Pa.	42,750	8	580	1	3.17
Casselman River at Grantsville, Md.	2,911	15	47	13	.22
Casselman River at Markleton, Pa.	37,300	10	98	7	2.76
Laurel Hill Creek at Ursina, Pa.	4,249	14	35	14	.31
Youghiogheny River at Connellsville, Pa.	85,110	5	64	11	6.30
Youghiogheny River at Sutersville, Pa.	184,400	3	108	6	13.66
Monongahela River at Braddock, Pa.	1,350,000	1	184	4	100.00

Blacklick Creek, Conemaugh River, Stonycreek River, Two Lick Creek, and Loyalhanna Creek accounted for 43 percent of the total sulfate load transported to the Allegheny River at New Kensington for 1980. The U.S. Department of Interior (1967) noted that the Kiskiminetas River discharged massive amounts of acidity (494 ton/d) into the Allegheny River, that the average manganese concentration was 23.2 mg/L, and that the pH ranged from 3.0 to 3.6. The study estimated about 485 mi of streams within the Kiskiminetas River Basin are affected by mine drainage.

More than 60 percent of the total sulfate load enters the Monongahela River above Greensboro, Pa., which is just downstream of the mouth of Dunkard Creek (table 2, fig. 3). The Monongahela River at Greensboro includes all the Monongahela drainage within West Virginia and the Dunkard Creek drainage area. About 68 percent of the sulfate load is not accounted for at Greensboro on the basis of the sum of the available WQN stations used in this study. Of the 853,300 tons of sulfate transported past Greensboro for the 1980 water year, only 270,000 tons are accounted for by summing all of the available WQN stations upstream. Thus, a significant amount of sulfate enters the Monongahela River either directly from mine discharges or from streams not included in the WQN sites. The U.S. Department of Interior study (1967) noted that streams severely affected by mine drainage enter the Monongahela River above Greensboro and include the West Fork River (acid load of 156 ton/d) and the Cheat River (acid load 204 ton/d). Smaller tributary streams between Fairmont and Morgantown, such as Buffalo Creek, Paw Paw Creek, Scotts Run, and Deckers Creek, contribute significant amounts of mine drainage to the Monongahela River. Downstream of Greensboro, Redstone Creek and South Fork Tenmile Creek contribute a combined total of 54,120 tons of sulfate, or about 4 percent of the total sulfate load to the Monongahela River (table 2). The Youghiogheny River, which enters the Monongahela River about 10 mi upstream of the Allegheny and Monongahela River confluence, contributed 184,400 tons or 14 percent of the sulfate load to the Monongahela River (table 2).

To compare sulfate loads transported by rivers and streams with different drainage areas, the sulfate load was normalized to the drainage area (table 2, fig. 5). The result is a sulfate yield, expressed as tons per square mile. The Monongahela River Basin had a sulfate yield of 184 ton/mi^2/yr compared to 105 ton/mi^2/yr for the Allegheny River Basin. Redstone Creek, Blacklick Creek, Conemaugh River, Buffalo Creek, Stonycreek River, Two Lick Creek, Dunkard Creek, and Loyalhanna Creek had the highest sulfate yields of all tributary streams in the study area (table 2). These basins have been extensively mined. Brokenstraw Creek and Conewango Creek, which are outside the coal extent (fig. 5) and contain no coal

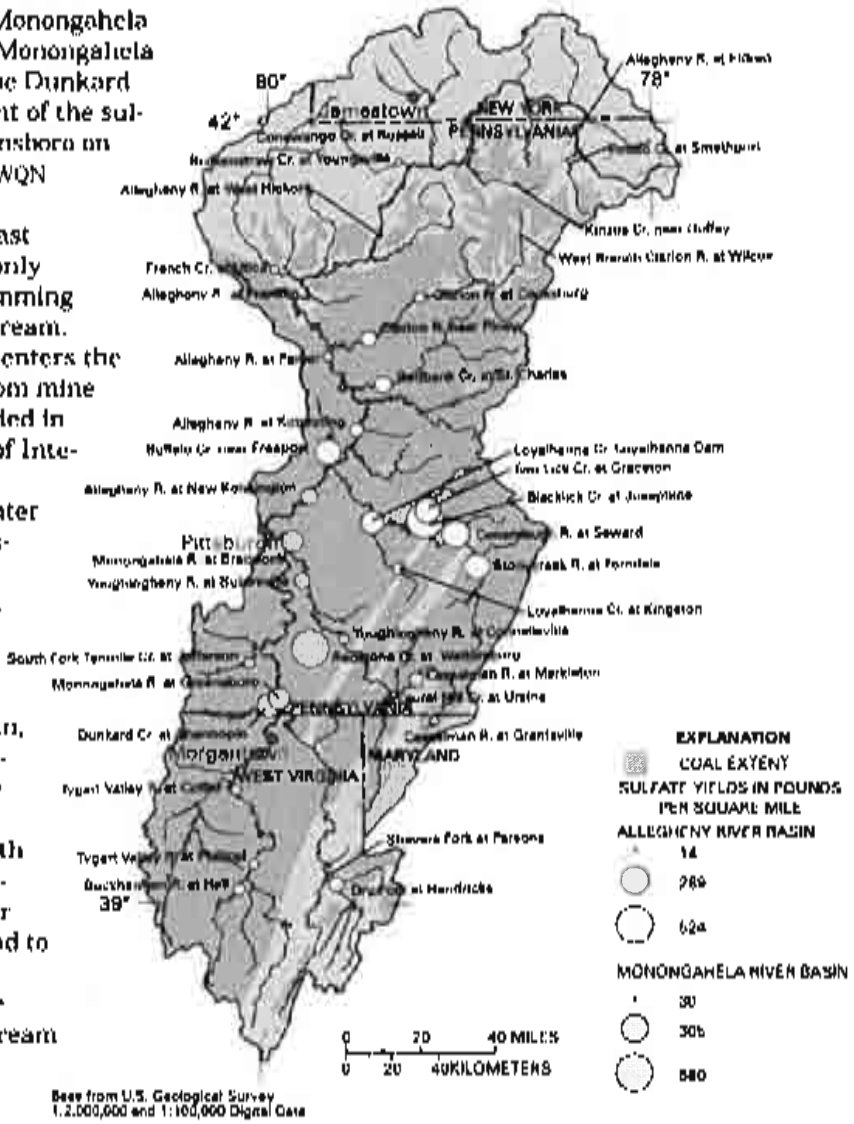

Figure 5. Sulfate yields at water-quality-network sites for the 1980 water year.

mines, transported sulfate yields of 25 and 24 ton/mi^2/yr, respectively. In a 1967 study, Blacklick Creek was identified as being one of the most mine-affected streams in the Allegheny River Basin with a stream pH below 3.0 (U.S. Department of Interior, 1967). The same study also identified Stonycreek River, Two Lick Creek, Conemaugh River, Loyalhanna Creek, Redstone Creek, and Dunkard Creek as streams severely affected by mine drainage.

SULFATE TRENDS (1965 - 1995)

A more detailed analysis was conducted on 15 of the 37 sites listed in table 3 to examine long-term trends in sulfate and their relation to coal mining. The sites were selected on the basis of the availability of long-term (1965-95) sulfate concentration and streamflow data. A Geographic Information System (GIS) was used to evaluate the extent of mining activities and mining effects in each of the 15 basins. A summary of available GIS data for the basins is shown on table 3.

The greatest range and largest maximum concentration of sulfate were in samples collected from streams in basins with extensive coal mining (fig. 6). The largest median concentrations of total sulfate were in samples collected from the mined basins of Blacklick Creek, Dunkard Creek, and Stonycreek River at 241, 234, and 181 mg/L, respectively. Samples collected from the Monongahela River at Braddock had a median sulfate concentration of 110 mg/L for the period 1965 to 1995. This is almost twice as high as the median concentration of 60 mg/L for samples collected from the Allegheny River at New Kensington. On the basis of the available GIS data, approximately 6,564 coal mines have operated within the Monongahela River Basin compared to 2,537 coal mines in the Allegheny River Basin. The Monongahela River Basin contains 2,685 abandoned mines and the Allegheny River Basin has 910 abandoned mines. Coal mining and AMD have resulted in 1,071 mi of streams classified with a no-fish designation in the Monongahela River Basin, and 1,320 stream miles so designated in the Allegheny River Basin (table 3).

The Estimator model (Cohn and others, 1989) was used to estimate trends in sulfate concentration through flow and seasonally adjusted raw sulfate concentrations. In this section of the report, trends in sulfate concentration refer to trend in "flow and seasonally adjusted" sulfate concentration, which is synonous with trends in sulfate loads as far as the magnitude and direction. Trends were

Table 3. Data extracted from Geographic Information System databases on coal mining for trend sites in the Allegheny and Monongahela River Basins

Site name	Median sulfate concentration in 1965-95 (milligrams per liter)	Pre-1950 coal mine sites within basin[1]	1950-95 coal mine sites within basin[2]	Number of abandoned mine lands within basin[3]	Total number of stream miles with no-fish designation[4]
Allegheny River Basin					
Allegheny River at Eldred, Pa.	12	0	0	7	13
Conewango Creek at Russell, Pa.	20	0	0	0	0
Brokenstraw Creek at Youngsville, Pa.	19	0	0	0	0
Allegheny River at West Hickory, Pa.	16	0	2	10	13
Clarion River at Cooksburg, Pa.	38	48	31	27	165
Redbank Creek at St. Charles, Pa.	94	66	72	71	178
Allegheny River at Kittanning, Pa.	36	351	259	405	864
Stonycreek River at Ferndale, Pa.	181	400	87	107	101
Blacklick Creek at Josephine, Pa.	241	207	12	31	43
Loyalhanna Creek Loyalhanna Dam, Pa	136	78	13	67	40
Allegheny River at New Kensington, Pa.	60	1,982	555	910	1,320
Monongahela River Basin					
Dunkard Creek at Shannopin, Pa.	234	49	22	4	2
Cassselman River at Markleton, Pa.	90	81	40	89	99
Youghiogheny River at Sutersville, Pa.	76	379	128	276	226
Monongahela River at Braddock, Pa.	110	5,958	606	2,685	1,071

[1] Data from U.S. Office of Surface Mining.
[2] Data from U.S. Geological Survey.
[3] Data from U.S. Office of Surface Mining.
[4] Data from U.S. Environmental Protection Agency.

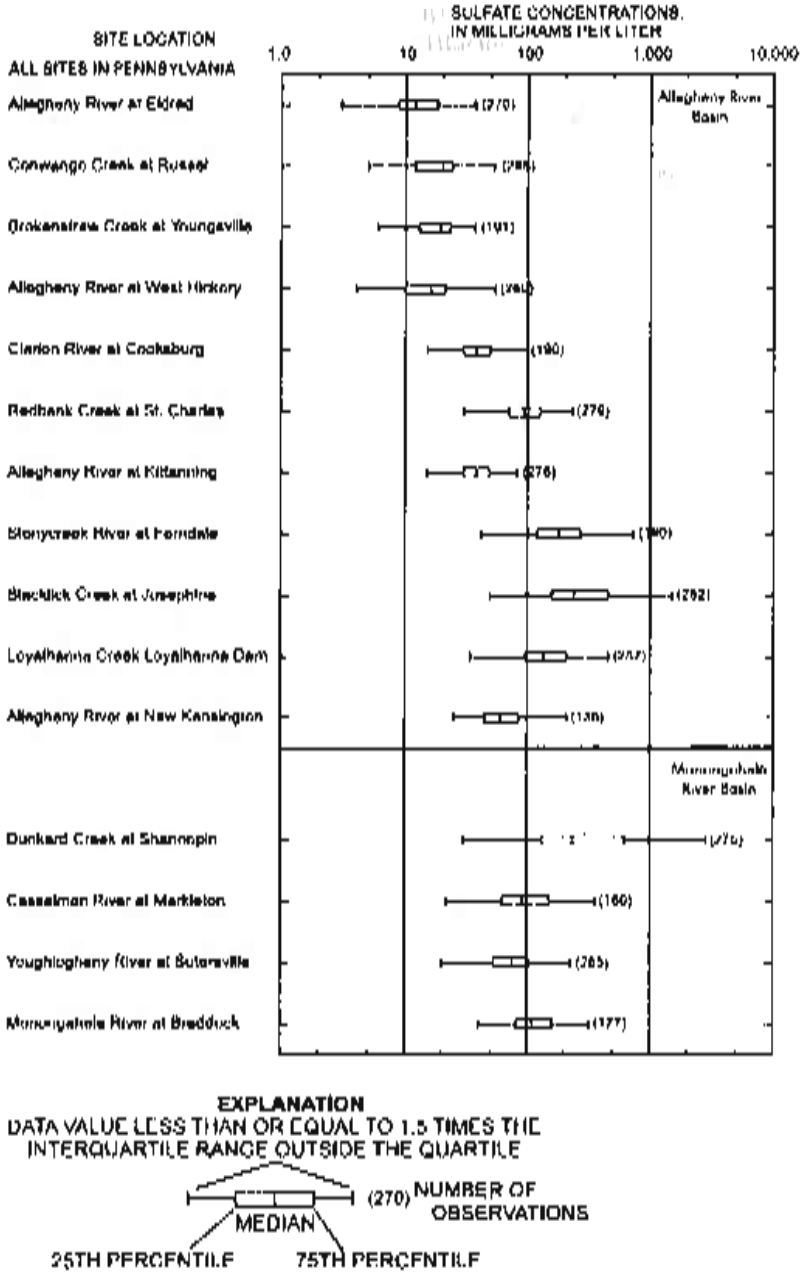

Figure 6. Distribution of sulfate concentrations for selected streams in the Alleghony and Monongahela River Basins, 1965-95

estimated for the period 1965 through 1995 for 11 sites in the Allegheny River Basin and 4 sites in the Monongahela River Basin (table 4, fig. 7). A p-value of less than 0.05 (table 4) indicates a significant trend (95-percent confidence level). The trend direction and magnitude (percent change in trend) are re-transformed from log space. The magnitude of change for the period is determined from the equation:

$$|\exp(B3\ t) - 1) * 100. \quad (1)$$

where B3 is the DECTIME regression coefficient from the Estimator model and t is the number of years for the time period examined.

Within the Allegheny and Monongahela River Basins, seven sites showed statistically significant trends in sulfate concentration between 1965 and 1995. For four of the sites, the sulfate trend is upward; for three of the sites, the trend is downward.

At two of the sites, Dunkard Creek and Stonycreek River, the trends appear to be related to increases in coal production in the basins from 1965 to 1995 (fig. 8). Coal production was estimated from county-based data for each basin (Pennsylvania Coal Association, 1995; West Virginia Office of Miners' Health Safety and Training, 1997). Coal production in the Dunkard Creek Basin, which averaged about 6.3 million tons between 1945 and 1980, began to steadily increase to 13.7 million tons in 1995. From 1965 to 1995, sulfate concentrations increased 38 percent in the Dunkard Creek Basin. Stonycreek River Basin, which had a 31-percent increase in sulfate concentration, had coal-production rates increasing from a low of 2.1 million tons per year in 1960 to a high of 6.5 million tons per year in 1981. Coal production and annual sulfate loads in the Stonycreek River Basin are somewhat cyclical from 1982 to 1995; however, the overall trend is upward for the period 1965 to 1995.

Table 4. Trends in sulfate concentrations at water quality stations in the Allegheny and Monongahela River Basins, 1965-95
[Gray-shaded sites are significant at the 95-percent confidence level]

STATION NAME	DIRECTION OF TREND	PERCENTAGE CHANGE			P VALUE
		MINIMUM	PREDICTED	MAXIMUM	
Allegheny River Basin					
Allegheny River at Eldred		2	27	64	0.0655
Conewango Creek at Russell		-7	16	44	.1821
Brokenstraw Creek at Youngsville	Up	11	34	62	.0025
Allegheny River at West Hickory		-5	20	51	.1115
Clarion River at Cooksburg		-21	-9	4	.1365
Redbank creek at St. Charles		-3	19	46	.0903
Allegheny River at Kittanning	Up	3	23	48	.0273
Stonycreek River at Ferndale	Up	0	31	72	.0482
Blacklick Creek at Josephine	Down	-61	-50	-35	0
Loyalhanna Creek at Loyalhanna Dam	Down	-52	-41	-29	0
Allegheny River at New Kensington		-33	-14	9	.2037
Monongahela River Basin					
Dunkard Creek at Shannopin	Up	8	38	78	.0105
Casselman River at Markleton		8	12	36	.2338
Youghiogheny River at Sutersville		24	12	0	.0566
Monongahela River at Braddock	Down	-40	-24	-4	.0173

Blacklick Creek at Josephine and Loyalhanna Creek at Loyalhanna Dam show statistically significant downward trends in sulfate concentrations between 1965 and 1995. Blacklick Creek had a 50 percent decrease in sulfate concentration. Coal production in the Blacklick Creek Basin, which reached its peak at 3.5 million tons per year in the 1940's, decreased to less than 1 million tons per year by 1995. Loyalhanna Creek Basin, which had a 41-percent decrease in sulfate concentration, had coal production rates decline steadily from more than 1.5 million tons per year in the 1940's to less than 200,000 tons per year in 1995.

The sulfate concentrations of Brokenstraw Creek at Youngsville increased 34 percent over the 30-year period. Because Brokenstraw Creek Basin was not mined, atmospheric deposition is considered a possible cause of this increase. However, sulfate concentrations in precipitation in the Eastern United States from 1983 to 1994 indicate a downward trend. Lynch and others (1996) attribute this trend, along with increases in pH, to passage of the Phase I Clean Air Act Amendments of 1990 (Public Law 101-549). Even so, the northern part of the Allegheny River Basin receives some of the highest sulfate deposition in the United States (National Atmospheric Deposition Program, 1998).

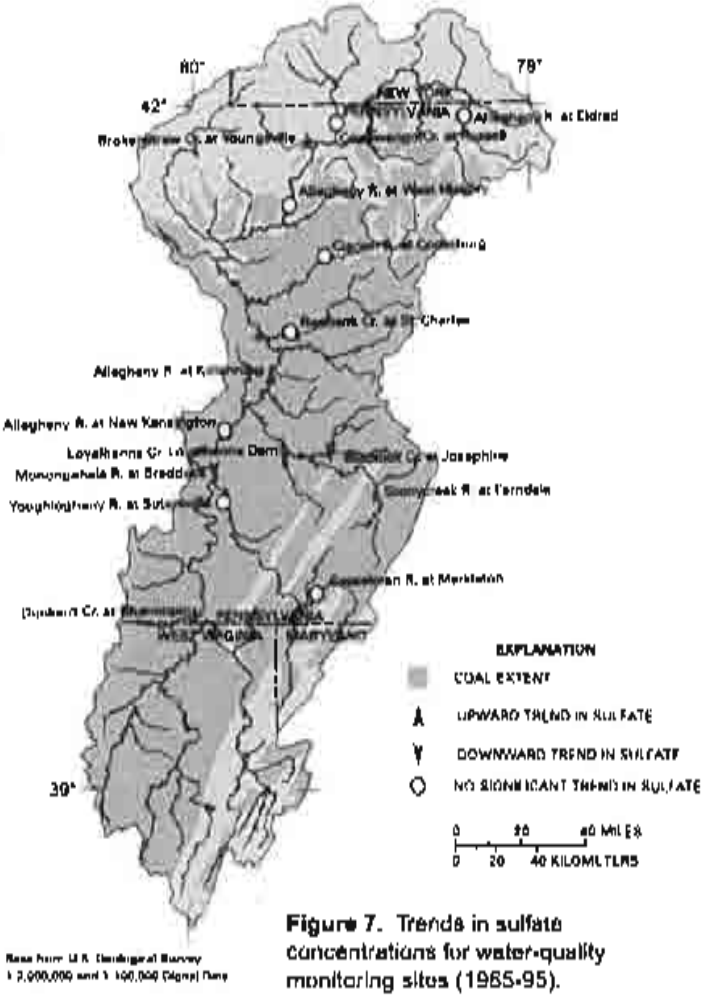

Figure 7. Trends in sulfate concentrations for water-quality monitoring sites (1965-95).

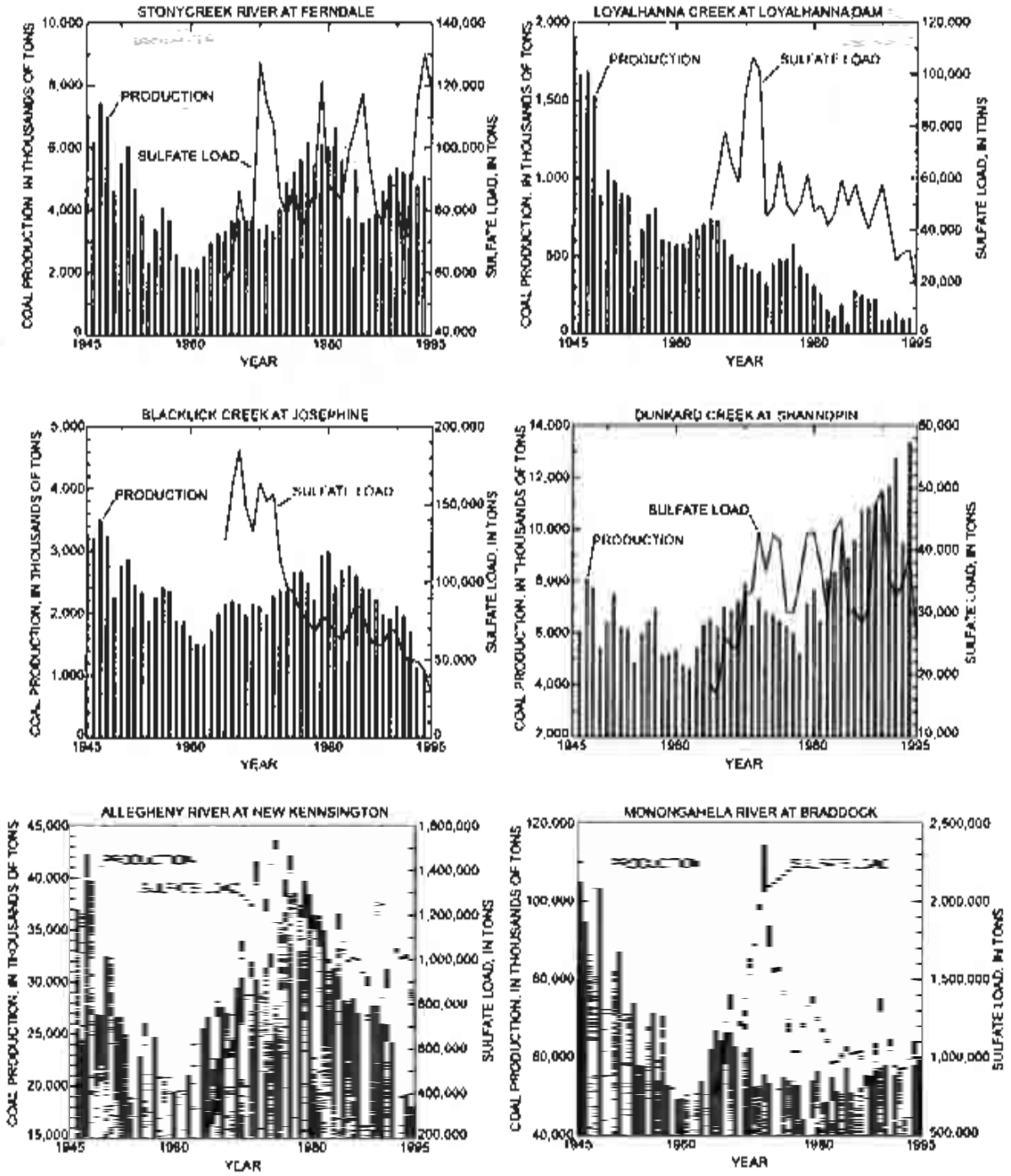

Figure 8. Estimated coal production and sulfate loads for selected coal-mined subbasins in the Alleghany and Monongahela River Basins.

Significant, but opposite, trends noted for the Monongahela River at Braddock and Allegheny River at Kittanning are more difficult to explain because of the large basin sizes—7,337 and 8,973 mi^2, respectively. Monongahela River at Braddock had a 23-percent decrease in sulfate concentration; Allegheny River at Kittanning had a 23-percent increase in sulfate concentration.

As the generation of mine drainage progresses through stages of initiation, propagation, and termination at individual mine sites and no new sulfide sources are exposed, concentrations of sulfate, acidity, and metals should decrease and eventually return to levels noted in unmined basins. Sulfate concentrations in the Loyalhanna Creek Basin have been steadily decreasing since 1950 (fig. 9). This trend is a function of decreasing coal production, reclamation of abandoned mine lands, mine-drainage treatment, and natural weathering of sulfide minerals. The flattening of the curve in figure 9 indicates further decreases in sulfate concentrations will occur slowly, and mine drainage will continue to affect water quality in Loyalhanna Creek for many years. This response should be similar for other streams and rivers affected by AMD in the Allegheny and Monongahela River Basins.

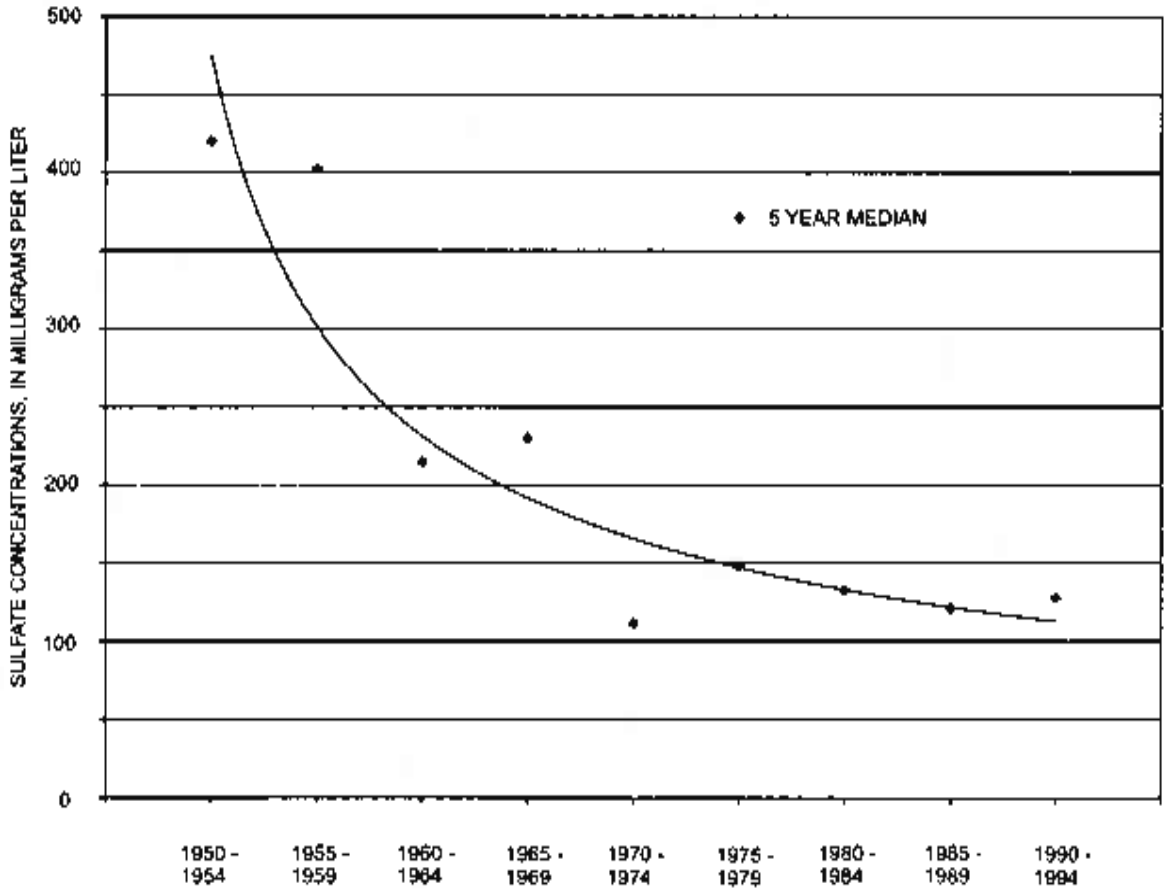

Figure 9. Median 5-year sulfate concentrations for the Loyalhanna Creek Basin, 1950-95.

SUMMARY AND CONCLUSIONS

Historical sulfate data for major streams and rivers within the Allegheny and Monongahela study unit of the NAWQA (National Water-Quality Assessment) Program were compiled to evaluate the status and trends of the effects of coal-mine drainage on water quality. Acid mine drainage (AMD) from coal mining has been identified as having the most widespread effect on water quality in the Allegheny and Monongahela River Basins. AMD discharging from deep mines and surface mines usually results in elevated concentrations of acidity, iron, manganese, aluminum, and sulfate in receiving streams and rivers. The environmental effects from 200 years of coal mining have resulted in 2,390 stream miles in the Allegheny and Monongahela River Basin that have been affected by acid mine drainage to the point of not being able to support fish communities. Most mine drainage emanates from abandoned mines that were in operation prior to the modern mining regulations established by SMCRA. Advances in the geochemical characterization of mine sites and improvements in mining technology have substantially reduced the number of recently permitted mine sites with poor post-mining water quality. This retrospective analysis represents a comprehensive effort to quantify these effects by using sulfate loads, yields, and trends as an indicator of coal mining in the study unit.

In 1980, the Allegheny and Monongahela Rivers transported a sulfate load of 1.2 million and 1.35 million tons, respectively, to the Ohio River at Pittsburgh. The Monongahela River Basin, although smaller in drainage area (7,337 mi^2), contributed 53 percent of the sulfate load to the Ohio River at Pittsburgh compared to the Allegheny River (drainage area 11,410 mi^2). Forty-three percent of the sulfate load for the Allegheny River at New Kensington came from Blacklick Creek, Conemaugh River, Stonycreek River, Two Lick Creek, and Loyalhanna Creek. These streams and rivers combine to form the Kiskiminetas River, which enters the Allegheny River about 15 mi below Kittanning. More than 60 percent of the sulfate load enters the Monongahela River above Greensboro, Pa.

Redstone Creek, Blacklick Creek, Conemaugh River, Buffalo Creek, Stonycreek River, Two Lick Creek, Dunkard Creek, and Loyalhanna Creek had the highest sulfate yields—from 580 to 196 ton/mi^2—of all monitored tributary streams. These basins have been extensively mined. The sulfate yields of Brokenstraw Creek and Conewango Creek, which are outside the coal extent and unmined, were only 25 and 24 ton/mi^2, respectively.

Monongahela River at Braddock had a median sulfate concentration of 110 mg/L for the period 1965 to 1995. This is almost twice as high as the median concentration of 60 mg/L for samples collected from the Allegheny River at New Kensington. Based on the available GIS data, approximately 6,564 coal mines have operated within the Monongahela River Basin compared to 2,537 coal mines in the Allegheny River Basin. The Monongahela River Basin contains 2,685 abandoned mines, and the Allegheny River Basin has 910 abandoned mines. Coal mining and AMD has resulted in 1,071 mi of streams classified with a no-fish designation in the Monongahela River Basin and 1,320 stream miles so designated in the Allegheny River Basin.

Flow and seasonally adjusted sulfate concentrations were used to estimate trends for the period 1965 through 1995. Sulfate increased 38 percent in the Dunkard Creek Basin and 31 percent in the Stonycreek River Basin. This trend appears to be related to increases in coal production. Blacklick Creek at Josephine and Loyalhanna Creek at Loyalhanna Dam show significant downward trends in sulfate between 1965 and 1995; these trends coincide with decreases in coal production. Blacklick Creek Basin had a 50-percent decrease in sulfate concentration; Loyalhanna Creek Basin had a 41 percent decrease in sulfate concentration. Sulfate concentrations in the Loyalhanna Creek Basin have been steadily decreasing since 1950. However, it appears that improvements in water quality (reduction in mine drainage constituents), which were rapid at first, show only gradual reduction as time increases. Consequently, it is likely to take many years for concentrations of metals, acidity, and sulfate to reach pre-mining concentrations.

The WQN stations maintained by Pennsylvania and West Virginia and the streamflow-gaging network of the USGS have provided data for evaluating the effects of mining on water quality in the Allegheny and Monongahela River Basins. Continued operation of this network will allow an opportunity to evaluate the effects of mining and also the effects of reclaiming and treating discharges from abandoned mine-land sites. This information will be useful in developing long-term trends and accounting for tributary effects on water quality in the Allegheny and Monongahela Rivers. The techniques used in this investigation also could be used in water-quality assessments of streams and rivers throughout the coal regions of Pennsylvania and West Virginia.

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EXHIBIT 15

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Hydrological changes following restoration of the Bois-des-Bel Peatland, Quebec, 1999–2002

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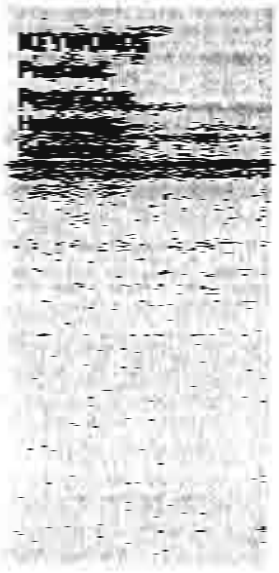

Summary Restoration efforts at peatlands drained for peat extraction attempt to return the hydrological conditions necessary for *Sphagnum* moss regeneration. One year of pre-restoration monitoring (1999) and three years of post-restoration monitoring were conducted at the Bois-des-Bel peatland near Rivière-du-Loup, Quebec, to evaluate hydrological changes that occurred at a managed section (restored site) relative to an adjacent abandoned section of the same peatland (unrestored site). Restored site evaporation was 74%, 74%, and 98% of unrestored site evaporation in 2000, 2001, and 2002, while runoff at the restored site was 83%, 35%, and 7% of unrestored site runoff over the same period. Seasonal soil volume change (swelling and shrinking of the peat) was evident at the restored site indicating rewetting of the peat profile despite the prolonged period between site abandonment and the initiation of the restoration measures (19 years). Site rewetting may have benefited from the relatively deep layer of residual peat (around 1.5 m). Higher seasonal mean water table and soil moisture were observed at the restored site relative to the unrestored site in 2000 through 2002 along with increased seasonal variability at the former location. Mean soil–water pressure was also higher than at the unrestored site and the range decreased, both spatially and on a seasonal and daily basis. The restoration techniques contributed to the recovery of hydrological conditions necessary for *Sphagnum* recolonization, though successful applications at different sites may be limited by specific peat and climate characteristics.

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Introduction

Successful large-scale restoration of mined North American peatlands has not been achieved. Early reports on the implications of restoration measures on hydrological (Price, 1996, 1997; Price et al., 1998) and ecological (Campeau and Rochefort, 1996; Rochefort et al., 2003; Lavoie et al.,

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2001) processes have greatly improved the understanding of the short-term response of relatively small-scale systems, ranging from plots to sections of peat fields.

Restoration procedures attempt to accelerate the return of the hydrological conditions necessary for *Sphagnum* moss recovery, specifically a high and stable water table (>−40 cm) (Schouwenaars, 1988), high soil moisture (>50%), and soil–water pressure above −100 mb (Price and Whitehead, 2001). The strategies employed must overcome the dramatic alterations in natural peatland hydrological processes caused by site preparation for peat extraction. Changes include the removal of living vegetation resulting in a bare, unstable peat surface (Price, 1997) and the installation of drainage ditches to lower the water table, which leads to increased subsidence (Schothorst, 1977; Hillman, 1992; Prevost et al., 1997), and alters water retention (Oleszczuk et al., 2000) and water transmission characteristics (Chow et al., 1992; Silins and Rothwell, 1998).

Seasonal plot-scale experiments have identified important techniques for improving site hydrology including the use of straw mulch for reducing evaporation (Price et al., 1998) and the blocking of drainage ditches to limit runoff (Price, 1997). Combining the work on improving hydrological conditions at cutover peatlands with details of *Sphagnum* regrowth has resulted in the development of a general approach for peatland restoration in Eastern North America (Rochefort et al., 2003), and have culminated in a system-scale restoration at Bois-des-Bel peatland, near Rivière-du-Loup, Quebec (GRET-PERG, 2003 (www.gret-perg.ulaval.ca)) prior to the 2000 growing season.

Initial site scale research at the Bois-des-Bel peatland has confirmed the importance of various aspects of the restoration approach. The addition of the straw mulch contributed to a 13% and 8% decrease in evaporation relative to an adjacent unrestored section in 2000 and 2001, respectively (Petrone et al., 2004b). The construction of bunds have also contributed to increased retention of snowmelt water which prolongs soil wetness during the summer, though bunds become ineffective at storing water once the water table drops below the peat surface (Shantz and Price, in press). At this site, higher soil moisture is also associated with greater density of straw mulch application (Petrone et al., 2004a).

Previous research at the Bois-des-Bel peatland has been valuable for helping to verify the importance of specific restoration measures, but the collective impact of these changes has not been fully evaluated. Identifying patterns in the hydrological characteristics is critical for evaluating the impact of restoration activities, and the potential for long-term restoration success. This includes the effect of new surface vegetation and moss cover (Groeneveld and Rochefort, 2002), loss of straw mulch by decomposition (Waddington et al., 2003a), and peat substrate changes through rewetting (von Waldow, 2002; Price, 2003). This research documents the hydrological changes by evaluating conditions prior to restoration (1999) and for three post-restoration years (2000 to 2002) to (1) identify how restoration techniques have influenced within-summer and year-to-year patterns of soil moisture, water table, soil–water pressure and peat volume change, and (2) consider factors that may influence the broad application of the current restoration approach.

Study site and restoration approach

Bois-des-Bel peatland is located approximately 10 km north of Rivière-du-Loup, Quebec (47°53′ N, 69°27′ W) at an elevation of 20 m asl. Climate data (1971–2000) (Environment Canada, 2003) recorded 5 km south (St. Arsene) indicates mean annual total precipitation is 962.9 mm (29% falling as snow) and the mean annual temperature is 3.2 °C. The entire ombrotrophic peatland covers an area of approximately 200 hectares. The peat is up to 3 m deep (though only around 1.5 m in the harvested area) and the site is underlain by a layer of marine clay that restricts vertical flow through the base of the peat deposit. Further details of site conditions and vegetation distribution within the undisturbed area of the Bois-des-Bel peatland are provided in Lavoie et al. (2001).

Approximately 11.5 hectares of the peatland were drained in 1972 for peat extraction and subsequently abandoned in 1980. The peat extraction area consisted of 11 fields numbered west to east (Fig. 1), each bounded by drainage ditches. Restoration measures were undertaken in the fall of 1999 on fields 1–8 (restored site, 8.1 ha) while field 9 was left as a buffer and fields 10–11 remained in their abandoned state for comparison purposes (unrestored site, 1.9 ha). At the restored site, vegetation and woody debris that had grown or become exposed in the time (Waddington and McNeil, 2002) since abandonment was removed. Old drainage ditches were blocked with well-decomposed peat approximately every 30 m along their length and a new collector ditch was dug along the south-east edge of the restored site perpendicular to the blocked ditches to allow monitoring of site runoff (see Fig. 1). Bunds were built perpendicular to the slope of the peat surface to block and store site runoff. Finally, *Sphagnum* diaspores collected at a nearby donor site were introduced, straw mulch was spread (4000 kg/ha), and a phosphorus fertilizer applied throughout the site (15 g/m^2) (Stephanie Boudreau, personal communication). For the purposes of this study, the "restored" site represents peat fields 1–8, for all years of study (i.e. even pre-restoration).

Methods

Monitoring occurred in the summer months (May–August) of 1999 through 2002. Specific monitoring start and end dates for all measured processes are listed in Tables 1 and 2. Evaporative fluxes were determined at both the restored and unrestored sites' micrometeorological stations (met stations) (Fig. 1) for all four annual monitoring periods. In 1999, the Priestley and Taylor (1972) approach was used. Soil lysimeters (see Van Seters and Price (2001) for details) were used to calibrate the Priestley and Taylor (1972) model. In all three monitoring years following restoration, the unrestored site micrometeorological instrumentation remained the same (the Priestley and Taylor approach) while restored site evaporation was determined using the eddy covariance approach. Set up details for both approaches are outlined in detail by Petrone et al. (2001) and Petrone et al. (2004a,b). Rainfall was measured at the restored site met station for all years with a manual rain gauge and

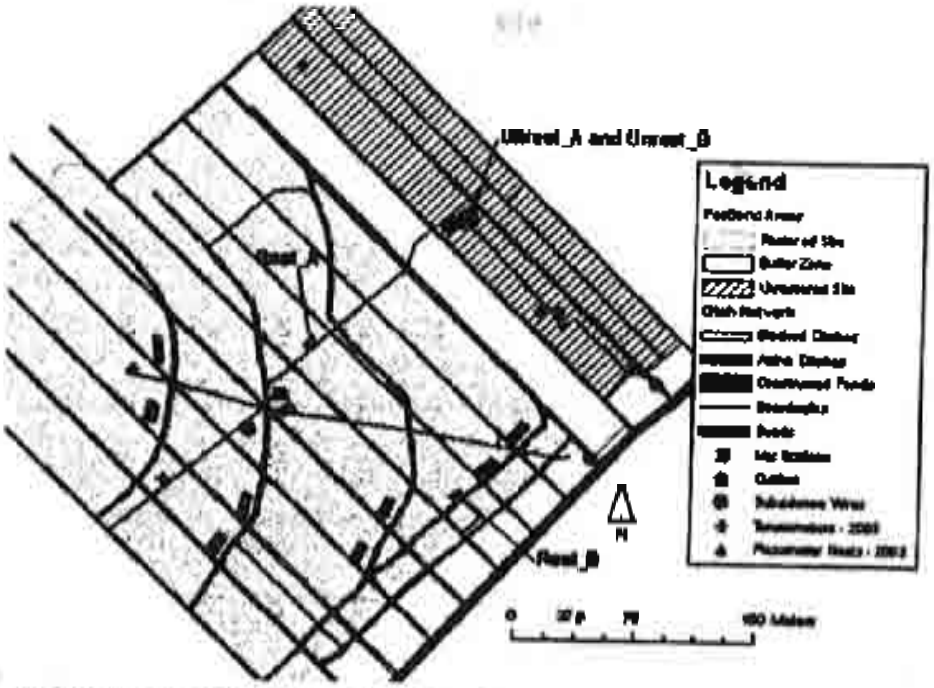

Figure 1 Map of the Bois-des-Bel peatland and location of hydrological monitoring locations for the 1999 through 2002 summer sampling periods.

Table 1 Precipitation, evaporation, and runoff at the restored (Res) and unrestored (Unres) sites of the Bois-des-Bel peatland for the summer (May to August) periods of 1999 through 2002

	1999		2000		2001		2002	
Monitoring dates (DOY)	178–243		130–243		131–243		138–243	
Site	Res	Unres	Res	Unres	Res	Unres	Res	Unres
Precipitation (mm)	142*		220		254		210	
Evaporation (mm)	164		248	334	374	501	253	257
Runoff (mm)	31	13	15	18	13	43	2	17

* Rainfall was measured from Day 152 in 1999 with total rainfall of 236 mm between Day 152 and Day 243. Evaporation was restricted to Day 178–243 and that is the precipitation component used in the analysis.

Table 2 Seasonal mean water table (wt), soil moisture at 5 cm depth (θ_{5cm}), and soil–water pressure at 5 cm depth (ψ_{5cm}) for the 1999 through 2002 summer periods at the Bois-des-Bel restored (Res) and unrestored (Unres) sites (seasonal standard deviation in brackets)

	Site	Monitoring period (DOY)	1999 μ (σ)	1999 n	2000 μ (σ)	2000 n	2001 μ (σ)	2001 n	2002 μ (σ)	2002 n
wt (cm)	Res	128–243	−68.4 (6.9)	48	−30.0 (9.5)	41	−30.4 (10.5)	95	−37.2 (14.3)	74
	Unres	128–243	−55.9 (7.4)	46	−45.5 (6.0)	35	−40.4 (6.4)	96	−44.3 (6.6)	72
θ_{5cm}	Res	130–243*	0.51 (0.03)	47	0.80 (0.06)	34	0.72 (0.03)	13	0.69 (0.09)	25
	Unres	130–243*	0.47 (0.02)	47	0.41 (0.03)	33	0.37 (0.01)	24	0.41 (0.04)	26
ψ_{5cm} (mb)	Res	140–242	−63.6 (26.9)	42	−6.8 (8.3)	36	−8.7 (9.7)	57	−24.8 (15.9)	45
	Unres	140–242	−45.7 (14.2)	42	−41.8 (17.3)	33	−29.8 (19.7)	58	−39.9 (16.8)	44

* In 2000, sampling was limited to days 170–243.

a tipping bucket rain gauge (Texas Electronics model TE525) logged every half hour.

The restored and unrestored sites represent unique watershed areas of 8.1 and 1.9 ha, respectively. Runoff was measured at both sites in all annual monitoring periods. Manual measurements were taken at the culvert (10 m length) outlet (see Fig. 1) using a calibrated bucket and stopwatch (average of three tests). Rating curves were developed using a flume in 1999 and 2000. The flumes were located just upstream of the culverts at both sites and stage was logged half hourly using a float and potentiometer connected to a Campbell Scientific™ CR10x datalogger. In 2001 and 2002, v-notch weirs were fastened directly to the outlet end of the culvert and the stage was measured using a Remote Data Systems (RDS) well. Relationships were developed between logged stage measurements (i.e. potentiometer and RDS wells) and manual measurements taken during discharge sampling.

Peat bulk density (ρ_b) was determined for all sampling years with the exception of 2001. Field samples of a known volume were weighed, oven dried, and re-weighed using standard methods of Klute (1986). Spatially distributed peat samples in the upper 50 cm were collected in cylindrical sample rings (volume 64.8 to 66.5 cm^3) either directly in the field or as subsamples from a Wardenaar™ peat corer (12 cm × 10 cm × 100 cm). In all cases, care was taken to ensure minimal compression during sampling. Seasonal peat subsidence and swelling was measured in the field at the restored and unrestored sites for all post-restoration monitoring years with rods anchored in the peat at 5, 10, 20, 30, 50, 100, and 150 cm depths, following the method of Price (2003). Readings of rod movement were taken relative to a level datum anchored in the underlying clay substrate at a frequency of 3–7 days in 2000 and 1–3 days in 2001 and 2002. Changes in peat volume for various layers were determined based on the movement of the anchors relative to the fixed datum. This method assumes volume change is occurring exclusively in the vertical direction which appears valid at the Bois-des-Bel site since there was no evidence of large scale vertical cracking in the peat.

Depth to water table (wt) was monitored manually every 1 to 3 days at the restored and unrestored met stations in all summers. wt was measured from the peat surface to account for any seasonal subsidence or swelling. Soil moisture (θ) was measured using the time domain reflectometry (TDR) technique (Topp et al., 1980). TDR probes were installed horizontally into the undisturbed wall of a pit, at depths of 2, 5, 10, 20, 30, and 50 cm. The pits, located near the two met stations, were then backfilled with peat. To ensure a solid position within the peat profile, probes were reinstalled at the beginning of each field season as the peatland thawed. Measurements were made using a Tektronix 1502B unit with waveforms saved to a laptop running the WATTDR program (Redman, 2000) and later used to determine the dielectric constant (Ka). θ values were calculated from the determined Ka measurements using the temperature dependent calibration function described by von Waldow (2002).

Soil–water pressure (ψ) was measured in all monitoring seasons at the met stations using straight and L-shaped porous cup tensiometers (Soil Moisture Systems, Tucson) installed at the beginning of each field season. Measurements were made manually every 3–5 days using a pressure transducer (accuracy of ±1 mb) (Soil Moisture Systems, Tucson). Measured pressure values were converted to actual pressure and total head (cm) based on the height of the water column in each tensiometer and its depth below the peat surface.

In 2002, more detailed spatial sampling of ψ occurred. Sampling locations are provided in Fig. 1 with equipment and sampling procedures the same as those described previously for the met station locations. Sampling frequency was every 3 to 5 days. In addition, intensive sampling occurred on Day of Year (DOY) 240 (a hot and dry day) starting at 06:00 and repeated every 2 hours until 18:00. Spatially distributed tensiometers (5 at each site) were measured at the 5 and 10 cm depths for both the restored and unrestored sites.

Horizontal saturated hydraulic conductivity (K_h) was measured weekly in 2002 at the restored and unrestored sites using the bail test method of Hvorslev (1951). While K_h measurements were also made in 1999 through 2001, the sample size was small and measurements were not used for interannual comparisons. Piezometers were emptied using a syringe and plastic tubing with recovery measured to a minimum of 80%. At the restored site, piezometer nests were located in the middle of an old peat field. Rest_A was located halfway between two blocked ditches in field 6 (see Fig. 1) while Rest_B was located 17 m into the peatland from the collector drainage ditch (see Fig. 1). At the unrestored site, piezometer nests were located close to the met station at a distance of 2 m (Unrest_A) and 8 m (Unrest_B) from the active north–south oriented drainage ditch (see Fig. 1). Piezometers were constructed from 2.5 cm internal diameter PVC pipe. Mean intake depths were 50, 75, 100, 125, and 150 cm (where the peat was deep enough) and the intake length was 15 cm. All piezometers were covered with Nitex screening (250 μm) to prevent clogging. Piezometers were inserted into pilot holes of similar dimension. Their elevation was surveyed relative to the site benchmark following restoration.

Results

Water balance components

Not including 1999 (shortened sampling period), 2002 had the lowest summer rainfall followed by 2000 and 2001 (210, 220, and 254 mm, respectively) (Table 1). Complete data are available for June, July, and August during all years monitored. In all years, total precipitation for June through August period was 85%, 66%, 82%, and 66% of the long term average, in 1999 through 2002, respectively. Following restoration, rainfall was below the monthly average for all summer months with the exception of May 2000 and July 2002. Evaporation (Table 1) was less at the restored site (which had straw mulch added as part of the restoration), in all post-restoration summers. In both 2000 and 2001, restored site evaporation was only 74% of that at the unrestored site. In 2002, evaporation at the restored site was 98% of that at the unrestored site.

Runoff was greater at the restored site (31 mm) than the unrestored site (13 mm) in 1999. After restoration, the unrestored site runoff was greater by 3, 30, and 15 mm, in

2000, 2001 and 2002, respectively (Table 1). For all post restoration summers, baseflow at both sites was negligible, particularly during dry periods (see Shantz and Price (in press) for more detail). The ratio of runoff relative to rainfall for the restored site decreased from 1999 through 2002 (0.186, 0.068, 0.051, and 0.010, respectively) while the unrestored site ratios were 0.078, 0.082, 0.169, and 0.081 for the comparable years.

Peat characteristics

Prior to restoration (1999), ρ_b in the 30–60 cm peat layer ranged from 0.101–0.161 g/cm^3 throughout the entire site (average = 0.130 g/cm^3, $n = 9$). Following restoration (excluding 2001 where no sampling was done), average ρ_b in the peat layer between 30 and 50 cm depth was 0.138 g/cm^3 ($n = 18$) at the restored site and 0.146 g/cm^3 ($n = 18$) at the unrestored site, both with a standard deviation of 0.013 g/cm^3. The 30–50 cm depth range was used to compare sites since there was less within season variability in density caused by shrinkage and swelling. A T-test ($\alpha = 0.05$) indicated ρ_b was not significantly different between the two sites for this depth range.

Following restoration, bulk density near the surface of the restored site varied as evidenced by seasonal changes in peat surface elevation. The restored site surface elevation dropped 4.7, 2.6, and 5.2 cm from its starting elevation in the 2000, 2001 and 2002 field seasons, respectively, while the unrestored site surface elevation change was less than 0.5 cm in each of the three years. Peat volume change at the restored site was strongly related to wt fluctuations with similar relationships for all post restoration years (Fig. 2). The greatest change occurred in the upper 30 cm zone where volume decreased 6.2%, 4.8%, and 9.6% in 2000, 2001, and 2002, respectively. In the layer between 30 and 150 cm, 1.9%, 1.1% and 2.4% change in volume occurred over the respective periods. In all three years, the greatest volume change with respect to wt in the upper 30 cm zone occurred in the early part of the monitoring period (prior to DOY 161) (Fig. 2) with the rate of change slowing later in the monitoring period (to day 243).

K_h was seasonably variable, decreasing during corresponding periods of low wt and greatest subsidence. Changes (monitored in 2002) were more evident in the −75 cm piezometers vs. those at −100 cm (Fig. 3). For example, K_h at Rest_B −75 cm piezometer decreased from 3.4×10^{-5} cm/s to 6.7×10^{-6} cm/s from day of year 142 to day of year 239. At the unrestored site, some variability in K_h associated with wt fluctuation was evident at the −75 cm depth but this was not associated with any measurable peat volume change.

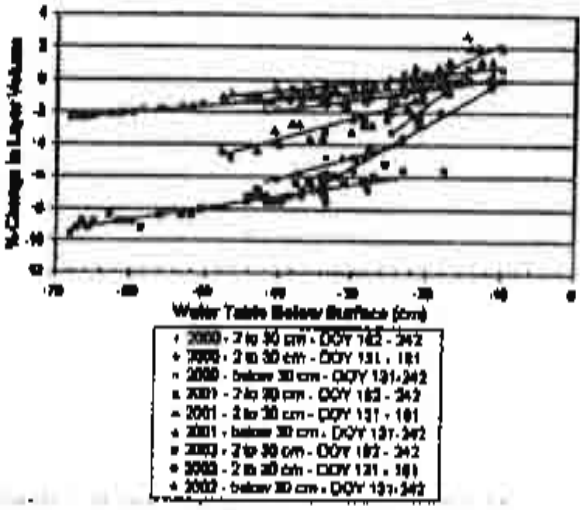

Figure 2 Restored site wt below surface vs. % change in layer volume in zones of 5–30 cm and 30 cm and below for the 2000, 2001, and 2002 summer periods.

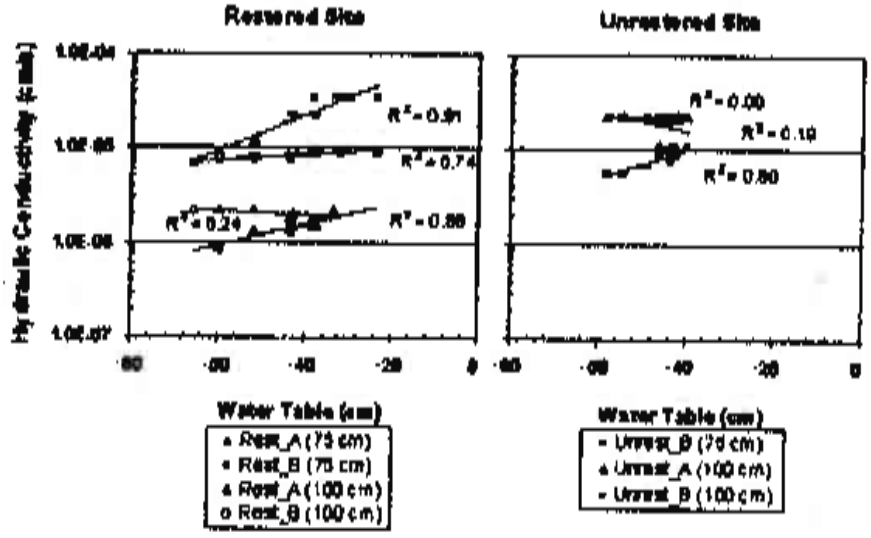

Figure 3 Changes in K_h relative to water table (wt) fluctuations during the 2002 summer field season.

Water table, soil moisture, and soil–water pressure

Prior to restoration, the mean wt depth at the restored site, and the unrestored site met station was −65.4 and −35.5 cm, respectively. Both sites showed little variability throughout the field season (Fig. 4 and Table 2). Following restoration, the wt in the restored site was closer to the soil surface than at the unrestored site, by an average of 15.5, 10.0 and 7.1 cm, in 2000, 2001 and 2002, respectively. However, the seasonal variability increased at the restored site, with standard deviation almost doubling from approximately 7 cm to 14 cm (Table 2). Mean wt was significantly closer to the surface in 2000 and 2001 but not 2002 (based on Mann–Whitney test and $\alpha = 0.01$).

Seasonal mean volumetric soil moisture at 5 cm ($\theta_{5\,cm}$) depth showed little variability at both the restored and unrestored sites in 1999 (SD of 0.03 and 0.02, respectively)

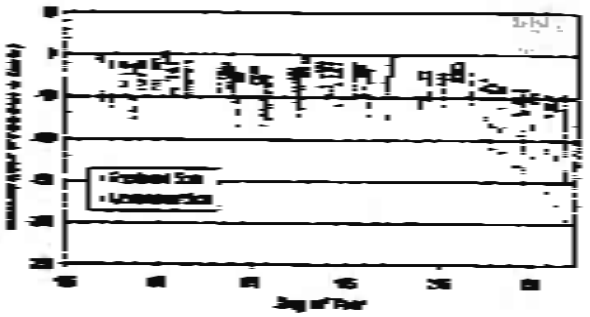

Figure 5 Seasonal soil–water pressure ($\psi_{5\,cm}$) trends at the restored and unrestored sites for the 2002 field season (each value represents daily average of all spatially distributed tensiometers at each site with error bars representing one standard deviation of all spacial samples).

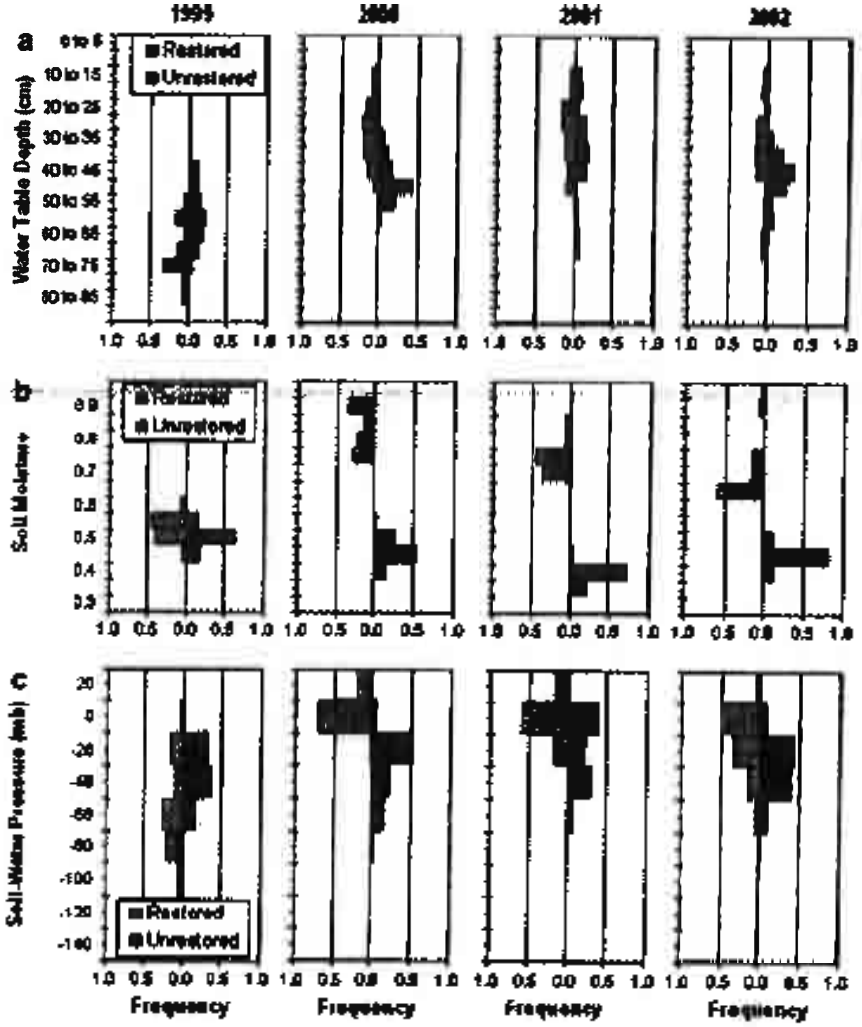

Figure 4 Frequency distributions during the 1999 through 2002 summer monitoring period for (a) restored and unrestored tower water table (wt) below surface, (b) restored and unrestored tower soil moisture at 5 cm ($\theta_{5\,cm}$), and (c) restored and unrestored met station soil–water pressure at 5 cm depth ($\psi_{5\,cm}$).

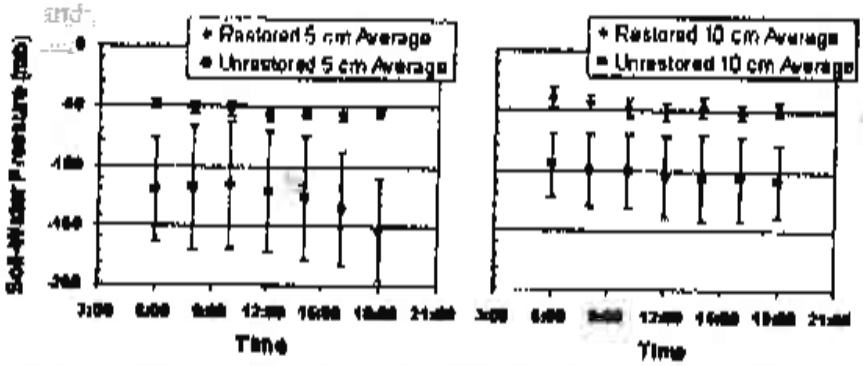

Figure 6 Daily soil–water pressure (ψ) trends at the restored and unrestored sites for 5 and 10 cm depths on Day of Year (DOY) 240 (error bars are one standard deviation).

(Table 2). In the post restoration summers, seasonal standard deviation remained relatively consistent at the unrestored site (0.03, 0.02, and 0.04 for 2000, 2001, and 2002, respectively) despite interannual differences in seasonal mean moisture content (Fig. 4). However, mean $\theta_{5\,cm}$ at the restored site was 40%–50% above that at the unrestored site. Summer restored site $\theta_{5\,cm}$ was consistently above 50%, while the unrestored site rarely had values at that level (the differences were significant in all post restoration years based on a Mann–Whitney test ($\alpha = 0.01$)).

In 1999, the mean soil–water pressure at −5 cm ($\psi_{5\,cm}$) at the restored site met station (i.e. in the pre-restoration year), was lower than at the unrestored site station (Fig. 4), and 3 of the 48 measurements during that year fell below −100 mb. Following restoration this trend reversed, and, $\psi_{5\,cm}$ at the restored met station site was consistently higher (Fig. 4) (Table 2). In all years, differences were significant based on a Mann–Whitney test ($\alpha = 0.01$). Furthermore, restored site $\psi_{5\,cm}$ never dropped below −100 mb following restoration and daily measured values were always above those found at the unrestored site. ψ in 2002 showed greater spatial variability at the unrestored site relative to the restored site for individual sampling days (standard deviation of 25.7 and 9.0 mb, respectively), as well as greater seasonal variability (Fig. 5). Spatially averaged $\psi_{5\,cm}$ fell below the −100 mb only at the unrestored site (Fig. 5). Two sampling locations used in the spatial average were below −100 mb for all sampling days following DOY 225 while the spatial average was below −100 mb for all but one sampling day following DOY 233. Greater spatial variability in ψ was also noted in diurnal measurements made in 2002 (Fig. 6). The unrestored site also had greater temporal variability in ψ relative to the restored site, particularly at the 5 cm depth where value ranges were 36.1 and 9.9 mb for the two sites, respectively. Restored site values decreased slightly towards noon but then stabilized while unrestored site $\psi_{5\,cm}$ continued to decrease throughout the afternoon.

Discussion

Water balance components

Water-balance components for the periods monitored indicated important hydrological changes occurred following restoration. Particularly, changes in storage (swelling and compression) within the peatland became a measurable component of the water balance with changes in dilation storage (as determined through volume changes below the water table) being equivalent to water losses from the site due to runoff. In all years, precipitation was the dominant water input (typical of ombrotrophic peatlands (Ingram, 1992)). However, the 1999 through 2002 summer periods were dry relative to the 30 year average. This was particularly the case for the three post restoration summers.

Evaporation dominated post-snowmelt water loss from the site with the unrestored site losing more water in all post-restoration years. In 2000 and 2001, total restored site evaporation was considerably less than the unrestored site. Petrone et al. (2004b) noted that the straw mulch affected temperature and albedo conditions near the surface of the restored site and likely reduced evaporation rates there. A comparable reduction in evaporation of a straw mulch covered peat surface was also reported by Price et al. (1998). Lower evaporation persisted at the restored site in 2002, although its effect was diminished because the mulch decomposed fairly rapidly following application in 2000 (Petrone et al., 2004b; Waddington et al., 2003a).

While three years may not be long enough to fully evaluate long-term runoff changes following restoration (Spieksma, 1999), this study shows distinct and immediate differences following restoration. Site runoff decreased in the years following restoration, both in terms of absolute magnitude and also relative to measured rainfall. The decreased runoff relative to rainfall at the restored site suggests that the restoration methods have been beneficial at retaining water on the site.

Evidence of site rewetting

Mean ρ_b in the 30–50 cm zone was slightly lower at the restored site (0.138 g/cm^3) relative to the unrestored site (0.146 g/cm^3) as expected based on the anticipated rewetting (and swelling) of the peat following restoration, although the difference was not significant ($\alpha = 0.05$). However, seasonal changes in peat volume observed at the restored site following restoration provide additional evidence of site rewetting. In the upper 30 cm zone, two distinct trends existed for each sampling year with the

greatest rate of volume change occurring early in the field season during the initial drop in the wt following snowmelt (Fig. 2). A similar finding was reported by Kennedy and Price (2005). Once the wt fell below 30 cm, shrinkage and secondary compression were occurring in the upper 30 cm zone (e.g. Price, 2003). Freeze–thaw cycles of saturated soils have been found to affect soil structure (Viklander, 1998) and this process is likely impacting the peat soil at Bois-des-Bel as well. The saturated peat conditions at the site prior to freezing (Shantz and Price, in press) resulted in an expansion of the peat beyond that which would occur simply from rewetting alone (Kennedy and Price, 2005). The rapid early season subsidence reflects this structural change.

Below 30 cm, peat volume changed at a consistent and low rate relative to wt fluctuations for all post restoration summers (Fig. 2). The similarity in patterns between years suggests that recovery of the volume-change function takes place immediately following restoration. However, the prolonged period between site abandonment and the initiation of restoration (19 years) may have reduced the potential for peat rewetting through increased decomposition and decreased mean pore size (e.g. Price, 1998; Price, 2003). Greater initial rewetting may be possible at sites where restoration measures are implemented immediately following abandonment (Kennedy and Price, 2004), although the relatively deep residual peat layer at the Bois-des-Bel site may have helped offset the potential impacts of prolonged decomposition.

Since the changes below 30 cm occur in generally saturated conditions, the results indicate changes in water storage through dilation storage. In 2000, changes in dilation storage represented as much as 21 mm of water during the driest period while 18 mm and 28 mm were lost from storage in 2001 and 2002, respectively. The estimates were calculated based on observed wt patterns and the slope of the wt–volume change relationship. When the wt is at its deepest point in the season, the change in saturated layer thickness reflects the change in water storage for that layer.

Seasonal changes in peat volume also influenced K_h at the restored site, decreasing during low wt periods (times of greatest subsidence). Similar field results were observed by Price (2003) in a partially restored peatland near Lac St. Jean, Quebec. At the unrestored site, negligible seasonal changes in K_h were observed at −100 cm and changes at the −75 cm level occurred in the absence of measurable seasonal peat volume change. However, given the rewetting of the restored site, and the consequent boost in soil methane production (Waddington et al., 2003b), the presence of bubbles that block pores (e.g. Kellner et al., 2004, 2005) may contribute to more seasonally variable K_h at the restored site.

While volume change has been shown to influence peat hydraulic characteristics due to change in pore structure (e.g. Chow et al., 1992; Sillins and Rothwell, 1998; Schlotzhauer and Price, 1999; Oleszczuk et al., 2000), the implications of these changes on restoration success are not clear. Price (2003) and Kennedy and Price (2004) have argued that seasonal changes in K_h may influence water transport to the surface, thus restricting vertical water losses when the wt is low and subsidence is greatest. However, since the anisotropic structure of the peat was not measured, the alteration of vertical water flows is not well understood. Further quantification of unsaturated zone K_h associated with peat volume change is also required, particularly for its importance in understanding differences in evaporation at restored and abandoned peatlands.

Differences in hydrological conditions between sites

Maintaining requisite conditions of wt, θ, and ψ is considered critical for the successful re-establishment of *Sphagnum* mosses (Price and Whitehead, 2001), particularly as a site initially undergoes the transition from mulch to vegetation cover. Measurable improvements in these conditions were observed at the restored site relative to the unrestored site.

Mean restored site wt was higher than it was at the unrestored site in all years following restoration, based on measurements made at the met station wells (with significant differences in 2000 and 2001). However, the range of wt fluctuation was also greater at the restored site than at the unrestored site, with periods where levels dropped below −40 cm at both sites (Fig. 4). An increase in the wt variability occurred at the restored site despite the moderating effects of peat volume change. The high wt variability at the restored site resulted from the enhanced snowmelt retention and flooding there (Shantz and Price, in press) which increased soil moisture and decreased available unsaturated pore space. Gilham (1984) noted the rate of wt change is strongly influenced by moisture and capillary fringe effects with small water inputs causing large changes in wt elevation. The deeper wt at the unrestored site, and the increased range of θ variability, indicates that the unsaturated zone at the unrestored site peat has more capacity to buffer wt changes than at the restored site which may contribute to the apparent hydrological disconnection between wt and surface exchanges when the wt depth exceeds 60 cm as hypothesized by Price (1997). There is a notable increase in wt at both the restored and unrestored sites, following restoration, which is difficult to explain. An average wt gradient of 0.008 between the restored site and the centre of the buffer zone suggests groundwater seepage may be occurring. However, the gradient between the centre of the buffer zone and the edge of the unrestored site is only 0.005 and with an average K_h value of 2.0×10^{-4} cm/s at the −100 cm depth at the Bois-des-Bel site, the influence was estimated to be less than 0.002 mm/day which did not result in improved $\theta_{5\,cm}$ and $\psi_{5\,cm}$ at the unrestored site.

$\theta_{5\,cm}$ was higher following restoration (Fig. 4). Despite interannual variability (which may partly be a reflection of reinstalling TDR probes each year), restored site values were consistently higher than at the unrestored site and were maintained above 50%, which Price and Whitehead (2001) argued as being important for sustaining *Sphagnum* diaspores spread during the site restoration. The greatest difference in θ between the two sites was near the peat surface, while there was very little difference beyond 20 cm depth. At the restored site, water could generally be supplied to meet evaporation demands while maintaining relatively high θ conditions. The straw mulch and vegetation cover along with ongoing shrinkage and swelling, helped to stabilize the plant water supply despite the increased

variability in wt. At the unrestored site, the soil was sufficiently moist to sustain high rates of evaporation, but the drier unsaturated peat has lower unsaturated K_h, thereby limiting upward transport of water from within the peat profile, causing further drying of the peat near the surface. Further evidence of this can be seen from the exaggerated diurnal change in ψ_{5cm}, compared to ψ_{10cm} (Fig. 6).

ψ_{5cm} was higher following restoration (consistently above −100 mb). However, Kennedy and Price (2004) have suggested that spatial variability in ψ values in abandoned peatland areas is much greater than generally accounted for in field monitoring. This was confirmed by the large spatial variability in ψ at the unrestored site (Figs. 5 and 6), particularly during prolonged periods with little precipitation. Restoration decreased spatial and temporal variability (Fig. 6) relative to the unrestored site despite greater wt variability.

Over the medium term, this study shows that θ and ψ improvements were maintained at the restored site despite decomposition of the straw mulch (Waddington et al., 2003a) and the transition to a vegetated surface. The return of vegetation (including vascular species such as *Eriophorum*) did not lead to increased drying at the restored site (see also Lavoie et al., 2005) although it may have contributed to the increased water table variability. Petrone et al. (2004a) have indicated spatial moisture patterns are related to surface vegetation cover. The role of vegetation recovery following restoration on spatial patterns in peat volume change, θ, and ψ, remains unknown.

Conclusions and recommendations

The restoration procedures applied at Bois-des-Bel peatland to enhance rewetting included the blockage of ditches and construction of bunds to retain precipitation inputs, principally snowmelt; and the application of straw mulch to reduce evaporation losses. During the construction of bunds (autumn 1999) the surface peat was reworked in part to level the surface between bunds, and partly to provide the material for bund construction. Shantz and Price (in press) reported that bunds were essential to retaining snowmelt water in the spring, but when the wt dropped below the surface, bunds provided no further hydrological function. Instead, during the summer period blocked ditches were generally effective at helping to reduce runoff losses from the restored site. Consequently, at the beginning of the growing season, the restored site had considerably more water available, and further runoff was curtailed by blocked ditches (Shantz and Price, in press). Petrone et al. (2004b) demonstrated how mulch reduces evaporation water losses from the restored site during summer. This study confirmed that evaporation from the newly restored zone was smaller than at the adjacent unrestored site, but also showed that the effect of the mulch had diminished almost completely by the third year after restoration. Nevertheless, with the additional snowmelt water retained, the net effect was to noticeably increase the summertime wt, and soil-moisture and water pressure in the near-surface zone.

This study showed that in addition to favourably adjusting the water balance components for rewetting the site, a series of other hydrological changes occurred that may be essential for successfully re-establishing an appropriate peatland plant community. Specifically, the volume-change function so important to water storage changes in many peatlands (Price and Schlotzhauer, 1999) was returned. With the ability to subside when water was lost (e.g. to evaporation), peat near the surface remained wetter relative to the unrestored site (Fig. 4). This compressible rewetted zone had higher ψ_{5cm} (Fig. 4), thus water remained readily available to plants, particularly *Sphagnum*, which relies upon relatively weak capillary forces (Hayward and Clymo, 1982) to draw water from the soil. The stark difference between the restored and unrestored site in this regard, and the similarity of the wt–volume change relationships in all years following restoration suggests that the volume-change function was restored quickly at the site.

The changes in K_h associated with changes in soil-volume (Fig. 3), are uncertain. Potentially, greater hydrological stability results if the potential for water loss diminishes (i.e. K_h decreases) as the system becomes drier. However, horizontal water losses appear to be a relatively small component of the summer water balance following restoration and it is likely that changes in K_h caused by volume-change are much less than changes caused by variable saturation (i.e. the unsaturated K_h function). Further research is warranted to quantify differences in unsaturated K_h at the restored and unrestored sites and the implications for influencing evaporation losses.

Another important change noted in the restored site was the homogenization of ψ distributions, both spatially and temporally. Whether a consequence of site levelling, or the general increase in surface wetness (assisted by the volume-change mechanism), spatial differences in ψ were significantly reduced, and the values were consistently higher. At the restored site the ψ remained uniformly above −100 mb, the pressure threshold beyond which the survival of *Sphagnum* diaspores may be severely limited (Price and Whitehead, 2001). The improvements in ψ (and θ) occurred despite the increased wt range at the restored site suggesting that wt alone is not a strong indicator of site suitability for *Sphagnum* recovery following restoration.

The potential for peat rewetting and its influence on hydraulic functions (i.e. K_h, ψ) is in part determined by the characteristics of the residual peat at the site to be restored. For example, the effectiveness of the volume-change process as a storage mechanism is related to the depth of remaining peat, and its state of decomposition (Kennedy and Price, 2005). At the Bois-des-Bel site, the length of time between abandonment and the start of restoration (19 years) may have decreased the full potential for peat rewetting but did not appear to largely affect the success of the restoration strategies. Waddington and McNeil (2002) estimated that 730 t of carbon was lost from the Bois-des-Bel research site prior to restoration. However, the relatively thick residual peat layer (≈1.5 m) provided an advantage for rewetting over more severely mined sites. Additionally, there was little harvesting (thus use of heavy machinery) on the site which may have decreased long term compaction.

Since site specific factors in substrate conditions, climate, and (post) harvesting conditions have the potential to dramatically influence the success of restoration, it is difficult to identify a specific restoration prescription for broad application. Testing of individual and linked restoration

measures under variable site conditions may only be possible with a mathematical model that fairly represents the processes and system characteristics. Despite this limitation, there are a number of general principles based on the results at the Bois-des-Bel site that have broader application in peatland restoration. The use of bunds and blocked ditches are important for retaining excess water during the snowmelt and spring period (Shantz and Price, in press). Given that the major water loss during summer is typically by evaporation, suppressing it with straw mulch is essential. Depending on the thickness of the peat deposit, and its ability to change its volume as it dries, sustaining wet conditions may be possible even after the mulch has decomposed. However, a second application of mulch may be necessary in situations where thin or rigid residual peat exists.

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EXHIBIT 16

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FOREST TO RECLAIMED MINE LAND USE CHANGE LEADS TO ALTERED ECOSYSTEM STRUCTURE AND FUNCTION

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Abstract. 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.

Key words: 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 Scheidiver 1986, Roberts et al. 1988, Johnson and Skousen 1995), high bulk density (Johnson and Skousen 1995, Burger and Zipper 2002), and poor soil texture

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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 physico-chemical 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 (Coward 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.

Table 1. Characteristics of the three study sites.

Characteristic	NEF	MAT-F	MAT-R
Slope (%)	9.9	4.5	4.5
Aspect	NW	SW	SW
Elevation (m)	690–775 m	815–850 m	775–835 m
Latitude	39°35′47″ N	39°35′39″ N	39°35′39″ N
Longitude	78°54′29″ W	78°53′29″ W	78°53′29″ W
Area (ha)	3.0	14.7	12.5
Land use	forest	forest	reclaimed mine grassland
Year of last disturbance	1981	1981	1985

Notes: NEF was a reference forested watershed whereas mining took place in MAT watershed. MAT-F refers to the portion that was clear-cut but never mined, and MAT-R refers to the section of the watershed that was mined and reclaimed. For all three sites, soils were Cookport series (very stony silt loam); mean air temperature was −8.6°C in January and 26°C in July; and mean annual precipitation was 113.2 cm. Long-term (1971–2000) mean monthly air temperatures and annual average precipitation were measured in Frostburg, Maryland, USA (Maryland State Climatologist's Office (http://www.atmos.umd.edu/~climate/)).

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.

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 (TNEFI). 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

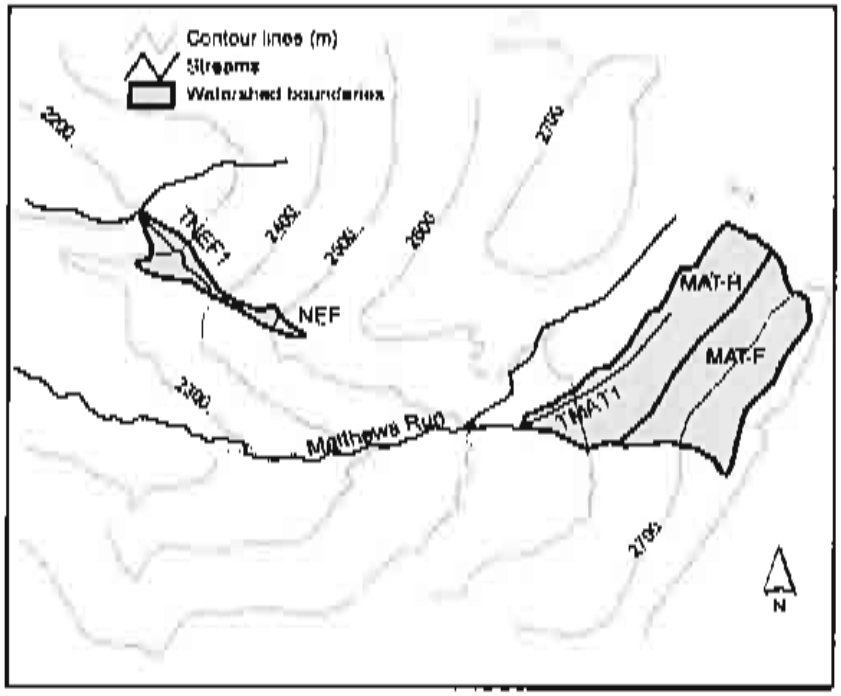

FIG. 1. Topographic map showing the relative locations of the watersheds of the study area in western Maryland, USA. TNEF1 is the stream that drains NEF watershed. TMAT1 is the stream that drains MAT watershed. TMAT1 is a tributary of Matthews Run. MAT is subdivided into a reclaimed mine section (MAT-R) and a forested section (MAT-F).

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).

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

TABLE 2. Soil characteristics of the three study sites.

Horizon and site	pH	Bulk density (g/cm^3)	%C	%N	C:N	Total P (mg/g)	Infiltration rate (mm/h)†	Thickness (cm)‡
Oe/Oa horizon								
NEF	4.21^{bc}	0.032^{a} (0.015)	32.9^{b} (6.6)	1.94^{b} (0.37)	16.9^{b} (1.1)	2.50^{b} (0.587)	300	6.4^{b} (0.81)
MAT-F	3.49^{ab}	0.068^{b} (0.015)	39.5^{b} (6.7)	2.33^{c} (0.24)	17.0^{b} (1.8)	1.81^{b} (0.734)	nd	5.7^{b} (0.98)
MAT-R	5.59^{c}	0.128^{c} (0.091)	15.5^{a} (0.52)	1.01^{a} (0.36)	15.8^{b} (1.4)	0.371^{a} (0.093)	<3	0.9^{a} (1.2)
A horizon								
NEF	3.99^{b}	0.845^{d} (0.211)	4.80^{a} (2.9)	0.328^{a} (0.183)	14.2^{a} (2.0)	1.24^{b} (0.758)	nd	nd
MAT-F	3.11^{a}	1.01^{e} (0.124)	11.1^{a} (1.2)	0.430^{a} (0.276)	25.2^{c} (2.3)	1.29^{b} (0.997)	nd	nd
MAT-R	5.15^{c}	1.26^{e} (0.152)	1.95^{a} (0.3)	0.110^{a} (0.025)	16.4^{b} (1.4)	0.375^{a} (0.165)	nd	nd

Notes: NEF is the forested reference watershed, MAT-F is the forested portion of the mined watershed, and MAT-R is the mined and reclaimed portion of the mined watershed. Mean values are presented with standard deviations in parentheses. Soil cores were collected at 3-m intervals along three 100-m transects. MAT-R soil cores did not always have an Oe/Oa horizon. $N = 45$, 10, and 40 for NEF, MAT-F, and MAT-R, respectively. Different letters within a column indicate significantly different means according to the Student-Newman-Keuls test ($P < 0.05$). Site and horizon factors were significant at $P < 0.01$ for all soil characteristics analyzed by two-way ANOVA. The interaction between site and horizon was significant at $P < 0.01$ for all soil characteristics except pH. An entry of "nd" indicates that no data were available.

† Steady-state infiltration rate. No statistical test was performed because exact values could not be determined for MAT-R.

‡ Differences among surface horizon thicknesses were analyzed using a one-way ANOVA followed by Fisher's LSD with site as the treatment (alpha = 0.05).

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).

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

TABLE 3. Soil temperatures in forested (NEF) and reclaimed mine (MAT-R) watersheds.

Site	Temperature (°C)	Mean daily range (°C)		Gravimetric soil moisture content (g water/g soil)
		April–October	December–February	
NEF	10.2 (1.15)	3.03** (0.33)	0.14** (0.18)	0.513** (0.34)
MAT-R	10.5 (0.64)	7.23 (2.80)	0.76 (0.24)	0.250 (0.06)

Notes: Values are means (and standard deviations) of three permanent plots each in NEF and MAT-R from 1 March 2002 to 28 February 2003. Gravimetric soil water contents of soil samples were used for buried-bag incubations from May through October 2000 ($n = 9$). Daily range refers to the difference between the daily minimum and the daily maximum temperature.
** $P < 0.01$, repeated-measures ANOVA.

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).

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.

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. 52000 and 47000 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 23900, 32900, and 62600 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.

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·ha^{-1}·yr^{-1}) and MAT-F (>3328 kg N·ha^{-1}·yr^{-1}) compared to MAT-R (1455 kg N·ha^{-1}·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,

TABLE 4. Number of fungal colony forming units and number of carbon sources utilized in soil cores taken from NEF and MAT-R.

Site	Fungal colony-forming units	Mean no. C sources utilized
NEF	79.2** (66.8)	5.0** (3.7)
MAT-R	60.1 (51.2)	8.7 (3.4)

Notes: Values are means (and standard deviations). Samples for fungal analysis were collected three times per year from 2001 to 2004 ($n = 211$). Carbon resource utilization determinations were restricted to samples from 2003 ($n = 18$).
** $P < 0.01$, nested ANOVA.

TABLE 5. C budgets for TNEF, MAT-F, and MAT-R watersheds.

Component	NEF	MAT-F	MAT-R
C pools (kg C/ha), aboveground			
Foliage	1500	1100	1200
Bole/branches	44 200	40 200	0
Twigs	6300	5700	0
Total aboveground†	52 000	47 000	1200
C pools (kg C/ha), belowground			
Oi	890	nd	300
Oe/Oa	6500	14 700	2000
Mineral soil	22 300	46 600	21 300
Fine roots	3210	1300	300
Total belowground‡	32 900	>62 600	23 900
Total C pools (kg C/ha)	84 900	>109 600	25 100
C flux (kg C·ha^{-1}·yr^{-1})			
Litterfall	1400	nd	1240

Notes: Total C pools are the sum of the weighted species specific average concentration. Average tree C content per species was calculated using measured tissue concentrations and published allometric equations for biomass. In MAT-R, C pools were calculated from quantitative harvests of 100-cm^2 quadrats. Soil pools were derived from core samples from three 100-m transects (NEF and MAT-R) or one 100-m transect (MAT-F) based on concentration and bulk density. An entry of "nd" indicates that no data were available.

† Excluding standing dead and coarse woody debris.

‡ Excluding coarse roots and coarse woody debris.

ranging from 2.12 to 3.55%, but legumes made up <1.5% of plant biomass (Ramsey 2002).

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·ha^{-1}·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.

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.

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).

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

TABLE 6. N budgets for TNEF, MAT-F, and MAT-R watersheds.

Component	NEF	MAT-F	MAT-R
N Pools (kg N/ha)			
Aboveground			
Foliage	72	64	2.5
Bole/branches	290	270	0
Twigs	58	54	0
Total aboveground	420	388	2.5
Belowground			
Oi	40	nd	2
Oe/Oa	390	860	130
Mineral soil	1650	2040	1310
Fine roots	150	40	10
Total belowground	2230	>2940	1453
Total	2650	>3328	1455
Inputs/outputs (kg N·ha^{-1}·yr^{-1})			
Wet N deposition	6.69	6.69	
Stream NO_3-N export	4.64	0.11	
Net retention of N	2.05	6.58	

Notes: Total pools are the sum of the weighted species-specific average concentration. Average tree N content per species was calculated using measured tissue concentrations and published allometric equations for biomass. In MAT-R, pools were calculated from quantitative harvests of 100-cm^2 quadrats. Soil pools were derived from core samples from three 100-m transects (NEF and MAT-R) or one 100-m transect (MAT-F) based on concentration and bulk density.

TABLE 7. Net N mineralization, net nitrification, and inorganic N concentrations (0–10 cm depth).

Site and horizon	Net N mineralization (mg N m^{-2} d^{-1})	Net nitrification (mg N m^{-2} d^{-1})	Net nitrification (%)†	NH_4-N concentration (mg/kg)	NO_3-N concentration (mg/kg)
NEF					
Oe/Oa	6.1 (6.3)	2.3 (2.46)	47.4 (31.5)	10.2 (10.6)	18.9 (43.7)
Mineral	54.7 (2.80)	60.0* (11.7)	83.7* (93.4)	33.3* (45.0)	7.1* (8.95)
MAT-R					
Mineral	27.1 (45)	2.3 (5.0)	23.7 (30.5)	7.1 (10.8)	0.3 (0.31)

Note: Values are the means (and standard deviations) of 32 and 48 samples for NEF and MAT-R, respectively, taken during the growing season of 2001.

† Excluding samples with negative net N mineralization values.

* $P < 0.05$ (nested ANOVA).

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.

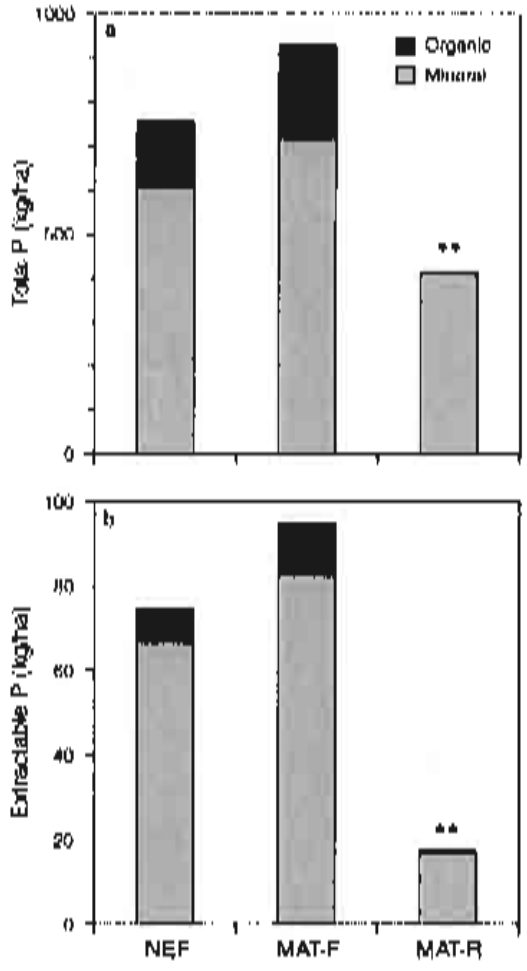

FIG. 2. (a) Total soil P pools and (b) bicarbonate-extractable (plant-available) P pools in a forested watershed (NEF), the forested portion of a mined watershed (MAT-F), and the mined and reclaimed portion of a mined watershed (MAT-R). Pools were measured 15 years after reclamation was completed to a depth of 10 cm.

** $P < 0.01$.

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*).

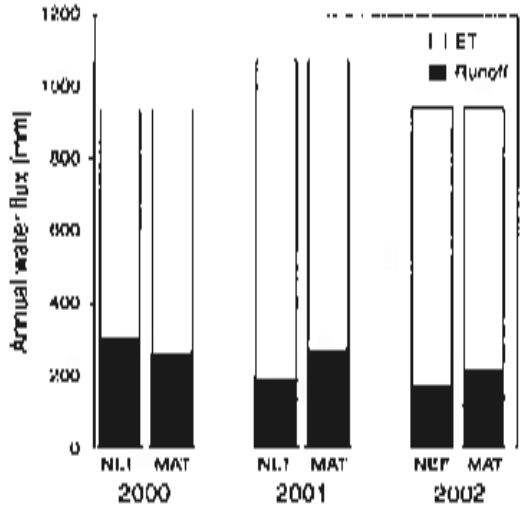

FIG. 3. Annual water flux (evapotranspiration [ET] and runoff) from NEF and MAT watersheds for water years ending in 2000–2002. The height of each bar represents annual precipitation.

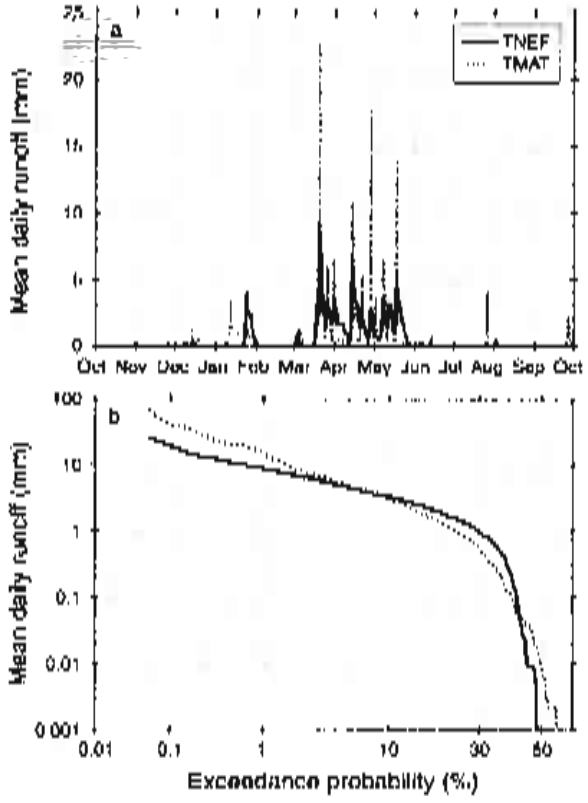

FIG. 4. Mean daily runoff of TNEF1 and TMAT1 (a) vs. time and (b) vs. exceedence probability (log–log scale), both for the 2000–2001 water year.

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).

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.

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.

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.

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

TABLE 8. Annual stream fluxes from a forested watershed (NEF) and a reclaimed mine watershed (MAT).

	Annual stream flux									
	Total P ($g \cdot ha^{-1} \cdot yr^{-1}$)		Particulate P ($g \cdot ha^{-1} \cdot yr^{-1}$)		DOC ($kg \cdot ha^{-1} \cdot yr^{-1}$)		NO_3–N ($kg \cdot ha^{-1} \cdot yr^{-1}$)		NH_4–N ($kg \cdot ha^{-1} \cdot yr^{-1}$)	
Year	NEF	MAT	NEF	MAT	NEF	MAT	NEF	MAT	NEF	MAT
2000	88.3	145	60.4	88.4	8.48	10.2	4.44	0.14	nd	nd
2001	71.8	167	48.1	99.1	5.46	17.0	4.16	0.13	nd	0.13
2002	nd	nd	nd	nd	1.21	11.9	5.12	0.05	0.01	0.02

Note: An entry of "nd" indicates that no data were available.

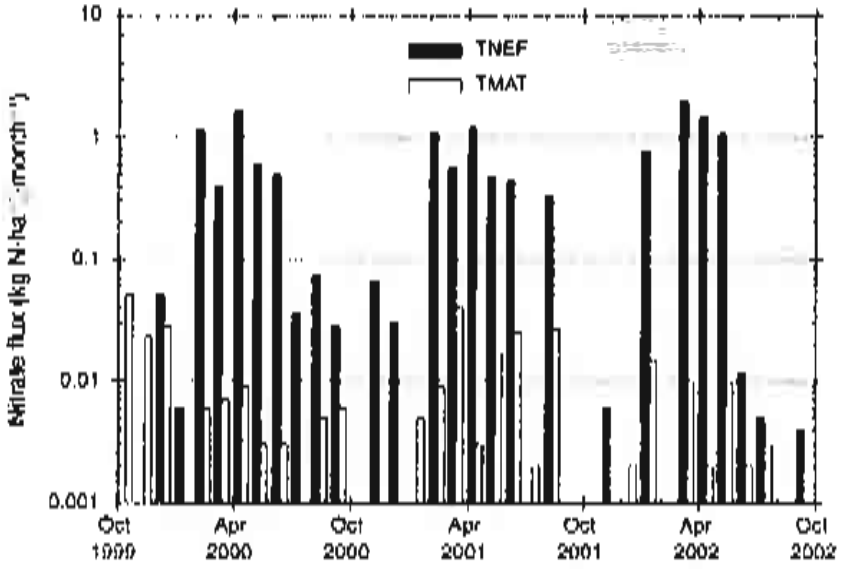

FIG. 5. Monthly runoff flux of nitrate from TNEF1 and TMAT1 from October 1999 through September 2002. Note the *y*-axis log scale.

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.

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, Bogner 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

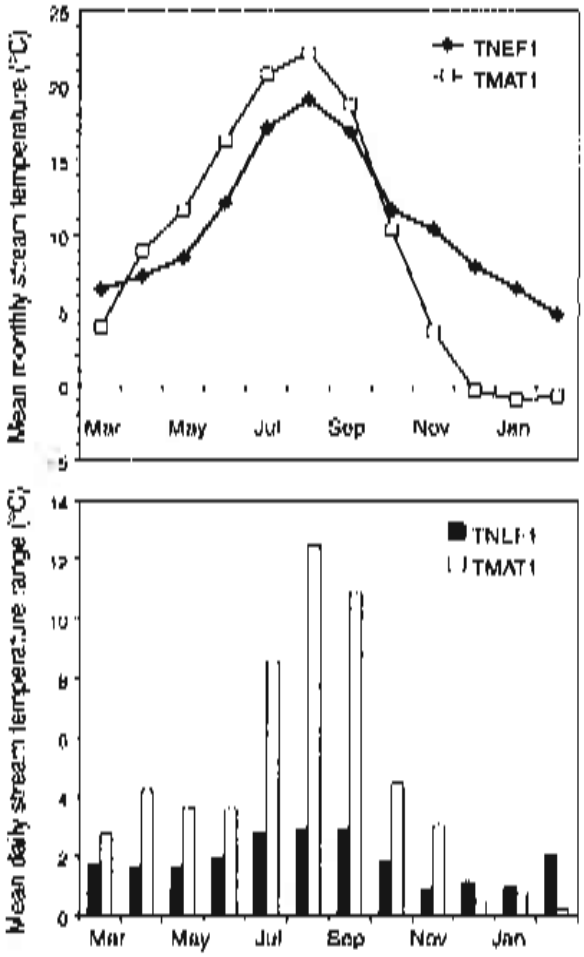

FIG. 6. Mean monthly stream temperature and mean daily range of stream temperature for TNEF1 and TMAT1 from March 2001 to February 2002.

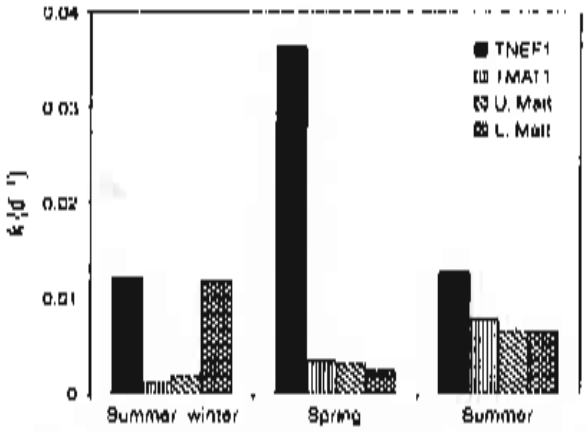

FIG. 7. Mean leaf decomposition rates (k) from a forested watershed (TNEF1), from a reclaimed mine (TMAT1), from the confluence of TMAT1 and Matthews Run (L. Matt), and from Matthews Run above the confluence (U. Matt). The summer–winter incubation period was July 2001 through March 2002 (228 days), the spring period was March through July 2002 (129 days), and the summer period was July through October 2002 (96 days).

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.

TABLE 9. Mean macrofaunal metrics at four locations on four sampling dates between 1999 and 2002.

Metric name	TNEF1	TMAT1	TMAT1–MATT confluence	Upper MATT
EPT (%)	79.0	6.3	58.5	54.1
E (%)	6.2	0.0	3.3	30.5
Tolerant organisms (%)	2.6	12.5	5.0	4.6
Dominant group (%)	47.7	25.0	35.9	30.5
Total no. organisms	386	16	329	627
Dominant group	Plecoptera	Coleoptera	Plecoptera	Ephemeroptera

Notes: EPT (%) is the percentage of Ephemeroptera, Plecoptera, and Trichoptera; E (%) is the percentage of Ephemeroptera. Samples were collected from a stream draining a forested watershed (TNEF1), from a stream draining the reclaimed mine site (TMAT1), from the confluence of TMAT1 and Matthews Run (MATT), and from MATT above the confluence. Tolerant organisms (%) refers to the percentage of individuals counted that had Tolerance Indices between eight and 10, indicating that they are relatively tolerant of pollution (Barbour et al. 1999). Dominant group is the family of macroinvertebrates that was most abundant at each site, and Dominant group (%) is the percentage of all individuals counted that were in that group.

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 microorganisms 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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EXHIBIT 17

Report on Chuitna Coal Project Aquatic Studies and Fish and Wildlife Protection Plan

Prepared by Lance Trasky
Lance Trasky and Associates

August 17, 2009

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Introduction

This report summarizes my review of the following list of documents related to the proposed Chuitna Coal Project.

(A) Aquatic Baseline: Existing Information for the Chuitna Coal Mine Project [Oasis Environmental, Inc.] (June 18, 2006);

(B) 2006 Freshwater Aquatic Biology Study Program [Oasis Environmental, Inc.] (April 4, 2007), including Appendices 1-8;

(C) 2007 Freshwater Aquatic Biology Study Program [Oasis Environmental, Inc.] (March 22, 2008), including Appendices 1-10 and Scanned Data Sheets;

(D) Wetland Functional Assessment [HDR Alaska, Inc.] (March 5, 2008);

(E) ASCMCRA Mine Site Permit Application, Part D7: Fish & Wildlife Protection Plan [PacRim Coal, LP] (July 2007), including all appendices;

(F) Agency correspondence concerning aquatic resources studies and study plans;

(G) 2008 Freshwater Aquatic Biology Study Program [LGL] [Jan. 2009].

My analysis of these reports focused on whether the aquatic resources studies were conducted using sound methodologies; whether the aquatic resources studies collected enough data and the right kinds of data; whether the aquatic resources studies provide an adequate understanding of the aquatic system; whether the reports provide an adequate foundation for a successful fish and wildlife protection plan; whether the fish and wildlife protection plan will successfully protect aquatic resources; and whether the aquatic resources studies provide adequate information for the U.S. Army Corps to make the required determinations under the 404(b)(1) Guidelines.

A. Aquatic biology: Existing information for the Chuitna Coal Project. Oasis Environmental 2006.

Purpose: to summarize existing information on freshwater aquatic biology collected in 1982, 1983, 1984 and 1987 by Environmental Research and Technology, Inc.

Analysis and Comments:

The existing information from the studies in the 1980's provided in the summary is very limited. It isn't clear if the original studies didn't collect a lot of the types of data now known to be necessary to assess the impact of the project or if Oasis didn't include everything in the summary. For example there is no information on phreatic and hyporheic groundwater flow investigations from the early studies.

Assuming that the 1982-1984 studies were equally rigorous, it would have been very helpful if the authors compared the data on fish and macro invertebrates to the data from the 2006-2008 studies in this summary. There are clearly differences in the number of adult Chinook, coho and pink salmon reported in the 1980's and the present studies. Some of the data summarized in Figure 4 Timing of Life History Events for Salmonid Species in the Chuitna Drainage also raises similar questions. For example, the table indicates that chum and pink salmon fry emerged and out migrated from mid-February to mid-May. Did this data come from age zero fry collected during winter sampling in the early 1980's? LGL did not report capturing any chum or pink fry in their winter sampling program in 2007-2008. As another example, the spawning period reported for coho from the 1982-87 studies as reported in Table 4 is September 1 through October 30. In 2008 LGL removed the weirs on Streams 2002-2003 and stopped counting adult coho on September 30. It is unclear whether this means that the timing of coho spawning has changed, that LGL missed a portion of the coho run, that the 1980's data is incorrect, or something else.

Summary: There are significant differences in the findings of the 1980's studies and the 2006-2008 studies. It is unclear if these differences are the result of changes in species composition, numbers and life history or differences in methodology.

B. Chuitna Coal Project 2006 Freshwater Aquatic Biology Study Program. April 4, 2007. Prepared by Oasis Environmental, Inc.

Oasis' study objectives related directly to fisheries:

1. Relative abundance, community composition, juvenile and adult distribution and spawning habitat
2. Water quality at trap locations and physical habitat evaluation locations
3. Channel morphology, habitat types, shape and substrate composition, and:
4. Human use of the fishery.

This information collected would be used to "assist with impact assessment" and "support project mitigation and reclamation plans. The channel morphology information would be "used to rebuild the affected streams to a level comparable with their original state and value."

Analysis and Comments:

Executive summary:

Page xii paragraph: "Adult salmon distribution was determined by aerial and ground surveys." Although aerial and ground surveys are often used to count salmon, these counts tend to underestimate the number of salmon present. There is within and between observer's bias in aerial and foot survey estimates of spawning abundance of salmon (Jones et al 1998 and Bue et al 1998). The ability to see salmon from an aircraft is dependent on many factors, including light level, time of day, skill of the pilot, water clarity and depth, and stream bank vegetation. Aerial and foot counts tend to significantly underestimate the number of salmon present (by 25% to 68%) and are useful only as general indices of abundance (Jones et al 1998). To accurately determine adult escapement in the Chuitna River drainage, a weir or sonar counter is also needed at the river's mouth. Counts should continue over 5 to 10 years because of the large natural fluctuations in population numbers and species composition over time.

2.2.1 Juvenile. Streams. Minnow traps: The 0.32 cm. mesh used to cover the minnow traps is large enough to allow age 0 salmonids to swim through it, so these fish may have not been accurately represented in the catch. Placing traps only in areas believed to be good juvenile salmon habitat likely biased the results.

2.5 *Paragraph 2*: If aerial surveys were impractical for estimating the number of adult salmon migrating up tributary streams due to poor visibility, and foot surveys were not used to estimate the number of spawning adult salmon, it is unclear how the estimates of adult spawning escapement in the tributaries were derived.

Ground surveys, Page 2.7. If lateral tributaries to streams 2002-2004 without minnow traps weren't surveyed, how did Oasis conclude that no adults spawned there? ADF&G

Habitat Division staff found both spawning adult and juvenile coho salmon in first and second order tributaries to the Hoholitna River in the 1980's.

2.5.1 Reference Reach Stratification page 2-14

2.5.2 Reach Selection Paragraph 3: Because of the great variation in stream morphology, stream flow, groundwater input , substrate, bank composition and other factors, it is not clear how a limited amount of data from a few hundred feet of stream channel in stream 2003 could be used to restore the 17.4 km miles of stream that will be excavated by mining. The report should also explain how information from streams 2002 and 2004 would be used in direct comparison to stream 2003 for restoration and rehabilitation purposes, since there appear to be substantial physical and biological differences between these streams.

It should be noted that Oasis also suggests that the data could be also used to rehabilitate streams 2002 and 2004 in the event that they are "inadvertently" impacted by mining. It seems likely that these streams will be also be impacted by mining to some degree. Studies from other strip mines show that mining within a watershed -- as will occur here in portions of the 2002 and 2004 watersheds -- affects surface water quality even when modern erosion and sediment control measures are used (Pond 2004). Mining through existing aquifers and pumping to dewater the mine could also affect groundwater flow in these drainages (National Research Council 1990, Schwartz and Crowe 1985, and Wilson and Hamilton 1978). However, it is not clear how a limited amount of morphological stream data could be used to restore streams affected by the loss of groundwater in the absence of adequate instream flow and groundwater data from streams 2002 and 2004.

Page 2.16-2.20: The use of the Rosgen stream classification system is useful to classify streams and stream reaches; however, there are problems with using classification data as a method to restore streams (Simon et al 2007 and Gillian 1996).

Page 2-20 paragraph 7. The flow of groundwater to streams 2002-2004, particularly during the winter, is one of the most critical factors in salmonid egg and overwintering survival (Baxter and McPhail 1999 and Douglas 2006). To successfully reconstruct a new stream that is equally as productive as the present stream 2003, it would be necessary to reconstruct a new shallow aquifer to provide the same amount of flow,

reestablish seasonal flow patterns and duplicate the existing chemical composition of phreatic groundwater,. This is important because: ground water in coal mine tailings often contains elevated levels of salts and metals such as zinc which is toxic to juvenile salmonids from 93 to 815 parts per billion, and ; spawning salmonids use chemical cues to locate their natal streams (Chapman 1978 and Dittman and Quinn 1996). I have searched the scientific literature and haven't found any instances where a new shallow aquifer has been recreated in a large drainage to support salmon spawning, or where premine ground water chemistry was duplicated. .

It is not clear how the flow of groundwater to streams 2002-2004 could be determined by measuring surface water temperatures and noting what appear to be seeps and springs at a few sites. First, groundwater may enter the hyporheic zone at one point but not surface for some distance downstream. In order to accurately measure and map groundwater input, it would be necessary to measure temperatures in the streambed systematically at many locations over the entire drainage. There may be more effective means available than thermometers to identify and measure groundwater input into Chutina drainage streams. A number of recent studies have used thermal infrared remote sensing to detect groundwater input into lakes and streams over a large area (Watershed Sciences 2007, Meijerink 1996, and Ackerman 2003).

Second, many species of salmonids select areas of groundwater input to spawn and overwinter (Bustard 1986, Baxter and McPhail 1999, Cunjack 1996 Swales et al 1986). Salmon eggs are deposited below the streambed and overwintering juvenile fish may move into streambed interstices in the hyporheic zone to escape ice and lethal low temperatures (Bjorn and Reiser 1991). Salmon eggs are deposited at some depth within the streambed. Large fish such as Chinook may dig as deep as 43 cm (17 inches) beneath the streambed and smaller fish 20-30 cm (8-11 inches). Phreatic groundwater at these depths would not be detected by surface water temperature measurements. In cold climates juvenile coho often select areas of groundwater upwelling to overwinter (Bustard 1986).

Stream habitat and abundance as CPUE page 3-11. The report indicates that: "Rosgen stream channel types will be used as the basis for river reconstruction after mining operations with each stream type encompassing a section of river with similar physical and biological attributes. Because of the importance of Rosgen stream channel types, statistical comparison of CPUE is based on Rosgen channel types." The report should explain why Rosgen stream channel types are so important and why they will be used to reconstruct streams after mining.

The problem with simply using the Rosgen classification system to reconstruct 17.4 km of a salmon producing drainage is that it doesn't take into consideration such things as water quality, surface water and groundwater seasonal flow patterns, stream temperature patterns, phreatic groundwater flow, and many other subtle factors that allow salmon to successfully complete their freshwater life cycle. Salmon in streams 2002-2004 are probably genetically adapted to the unique conditions in each of these streams and even small changes will likely result disrupt their life cycle.

Habitat spawning surveys. Page 3-35 paragraph 1. The range of temperatures reported at spawning redd locations (coho 7.32 to 11.36 degrees C, Chinook 11.60 to 15.56 degrees C, and pink 9.78 to 12.13 degrees C) is interesting. The lower temperatures at coho and pink redds could be an indication that coho and pink salmon are selecting sites with upwelling groundwater which in the summer would be colder than surface waters. Stream temperatures in the Cook Inlet basin tend to peak from May 15 to August 15 (Kyle and Brabets 2001). In 2006, Chuitna Chinook spawned from July to August, pinks

from mid-July to mid-August, and coho from August 15 to October 15 (Figure 4-1). However, it is not possible to determine from the data provided if salmon are selecting spawning areas with groundwater input, because water temperatures were taken at the surface of the redds instead of 8 to 17 inches into the stream bed where salmon eggs are deposited (Bjorn and Reiser 1991. Stream temperatures may also vary by several degrees daily and tend to be lower during pink and coho spawning.

Adult Distribution page 4-14: My experience is consistent with Oasis's finding that beaver dams do not obstruct adult upstream migration. Because coho, chum and pink salmon runs tend to coincide with late summer and fall rains when beaver dams overflow, salmon and other species of fish are usually able to move upstream around or over the dams with little trouble. I have observed coho, sockeye, chum salmon, white fish, and Dolly Varden moving upstream past beaver dams during high water events. I also agree that given current coho numbers, ERT's findings that "beaver dams were responsible for limiting adult coho access to far reaches of the drainages in 1982-1984" and that no coho were observed in stream 2003 in 1982, seem unusual. It could mean that there has been a major species shift from Chinook to coho salmon in stream 2003 or that coho's were present in the system but earlier studies weren't rigorous enough to detect them. Either scenario warrants additional research on coho and Chinook salmon abundance and life history. I discussed this with ADF&G research biologists and no major shift in relative coho and Chinook abundance was observed in other Cook Inlet streams from 1982 to 2009 (Hasbrouck 2009).

Juvenile fish habitat considerations, pages 4.6-4.7. Recent scientific studies have identified groundwater flow as a critical factor in the selection of adult salmonid spawning habitat, juvenile overwintering areas and the survival of eggs, fry and juvenile salmonids in cold climates (Leman 1993, Giannico and Hinch 2003, Malcolm et al 2004, Baxter and McPhail 1999, Lorenz and Filer 1989, and Douglas 2006). However, there is no discussion of groundwater flow as a habitat factor in the report. The 2006 study didn't identify sources of groundwater or investigate the relationship between phreatic groundwater in the hyporheic zone and salmonid spawning and overwintering in streams 2002-2004. Figures 1 and 2 illustrate how groundwater enters streams. Mining coal to a depth of 300 feet will remove all the layers of sediment or lithological units of low permeability that confine the aquifers (confining units) and currently provide shallow groundwater to stream 2003 and possibly streams 2002 and 2004. To restore fish habitat in these streams after mining it, it would be necessary to restore these confining units and provide the same quality and quantity of groundwater.

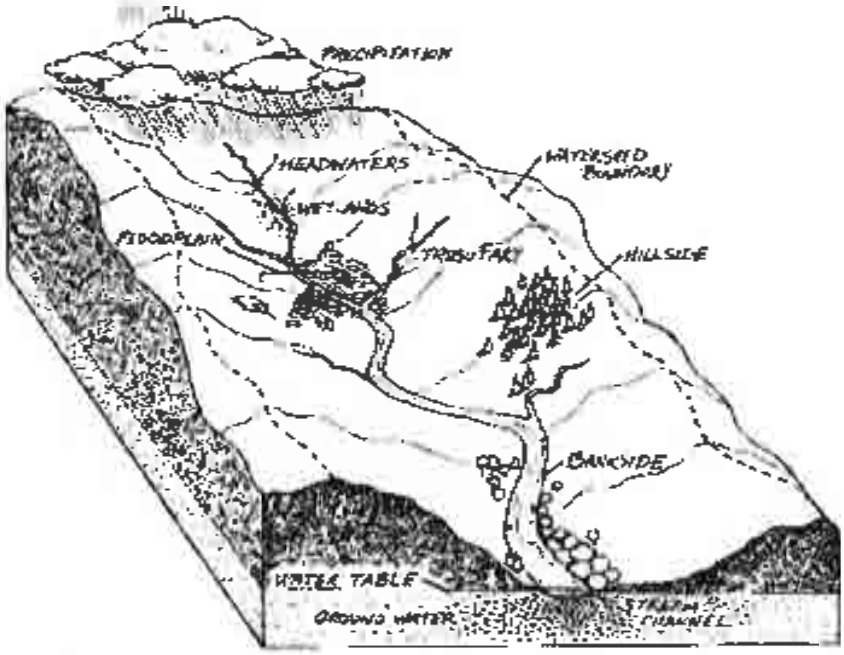

Figure 1: Cross Section of a Watershed (Source USEPA)

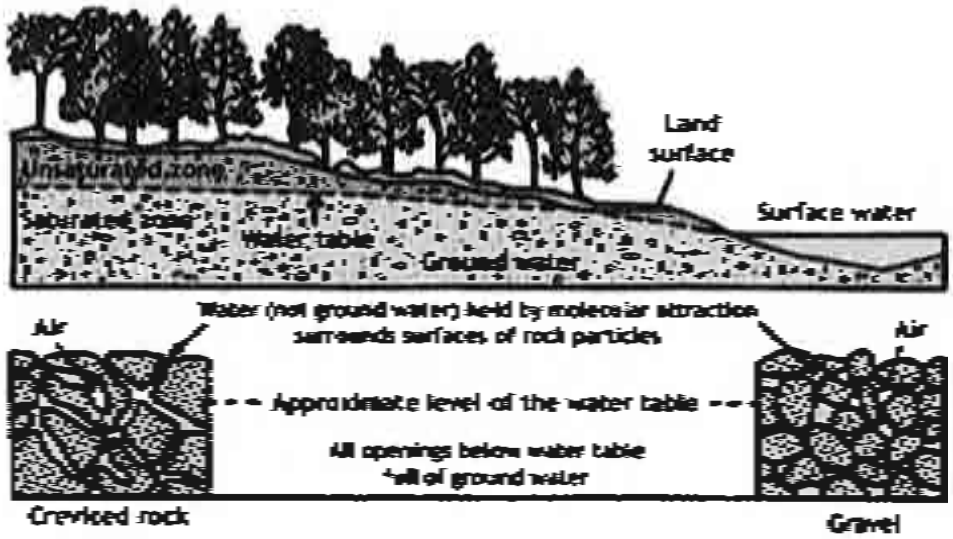

Figure 2: Cross Section of a Stream Drainage (Source: USGS)

Groundwater is critical because it maintains stream base flow and moderates water level fluctuations, particularly during the 5 to 6 months of winter when there is no liquid precipitation. It provides stable temperatures and thermal refugia for fish. It provides

water for riparian vegetation which controls bank strength and the rate of erosion (Douglas 2006). It also creates the hyporheic zone (Figure 3).

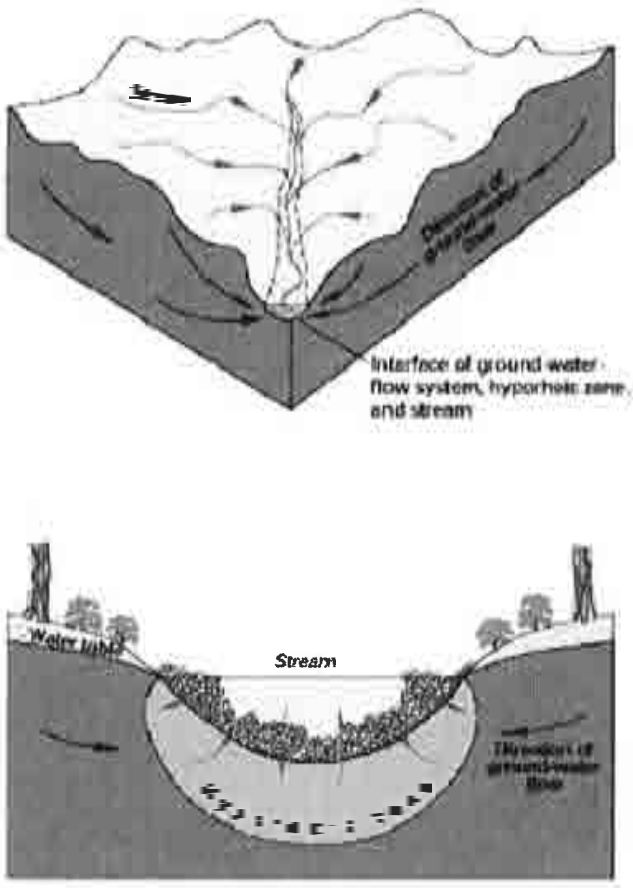

Figure 3: Hyporheic Zone (Source: USGS Circular 1186)

The hyporheic zone is the region beneath and lateral to a streambed where there is mixing of shallow groundwater and surface water. It is an active ecotone between the surface stream and groundwater. Exchanges of water, nutrients, and organic matter occur in response to variations in discharge and bed topography and porosity. Upwelling subsurface water supplies stream organisms with nutrients and cool water in the summer and warm water in the winter. Downwelling stream water provides dissolved oxygen and

organic matter to microbes and invertebrates in the stream bottom (Boulton et al 1998). Upwelling groundwater is vital because it protects salmonids and other cold water fishes from water temperatures that exceed their thermal tolerance in the summer (Hayes 2009). Groundwater provides overwintering habitat free of subsurface ice and protects fish eggs, larvae and juvenile fishes from freezing in the winter (Power et al 1999).

Two types of groundwater influence streams: Hyporheic groundwater and phreatic groundwater (Poole and Berman 2001). Hyporheic groundwater is from the alluvial material which underlies the streambed. It travels downstream along localized pathways before emerging further downstream. Phreatic groundwater comes from the catchment's aquifer and feeds a stream by entering the bottom of the alluvial material and mixing with the hyporheic groundwater (USGS 2006). Groundwater from the phreatic aquifer influences stream temperature when it enters the stream. The two-way exchange between the alluvial aquifer and the stream channel is perhaps the most important stream temperature buffer (Douglas 2006).

The hyporheic zone is an area of intense biochemical activity. Biogeochemical processes within the upper few centimeters of sediments have a profound effect on the chemistry of groundwater and surface water that mix in that area. Biogeochemical process is the partitioning and cycling of chemical elements and compounds between the living and non-living parts of a stream. The highly interactive nature of physical, chemical and biological processes in the hyporheic zone play a central role in the functioning of stream ecosystems (Malcolm et al 2003).

Streams can be classified as gaining, losing, or disconnected. Gaining streams gain surface flow from inflowing groundwater and losing streams lose flow through the streambed to the groundwater. Disconnected streams are streams where there is an unsaturated layer of air between the stream and groundwater. Streams can be both gaining and losing.

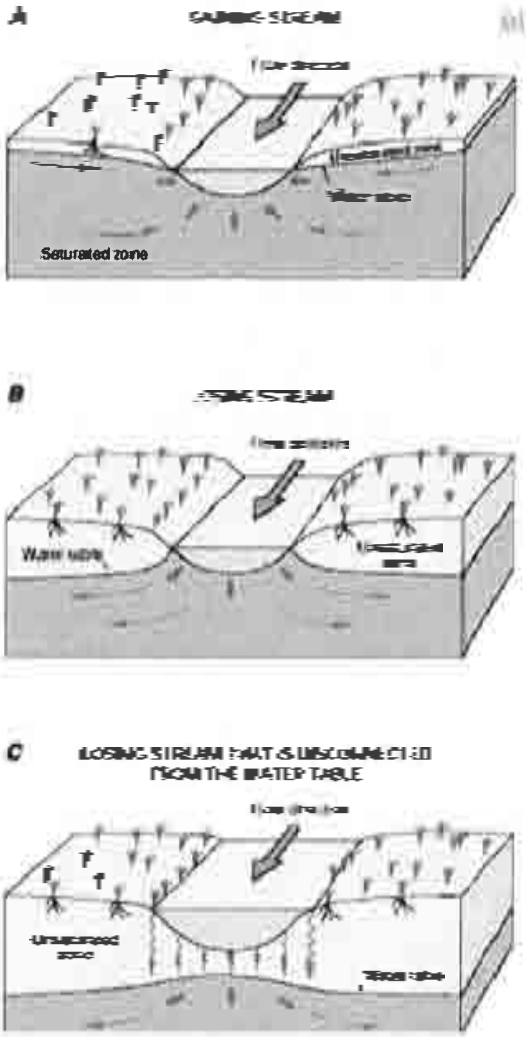

Figure 4. A. Gaining stream. B. Losing stream and C. Disconnected stream.
Source: USGS Circular 1135

Streams 2002-2004 appear to be gaining streams but could have sections which lose water to a deeper aquifer. However, it does not appear that PacRim or its contractors has collected enough data to make a stream segment by segment determination. Deep mining through the existing aquifers and pumping to draw down the aquifers and dewater the pit would likely lower the water table substantially. As a result, Stream 2003 would likely become a disconnected stream and lose water. Figures 4A, 4C, and 5. It is possible that streams 2002 and 2004 could also be affected by the draw down of the water table.

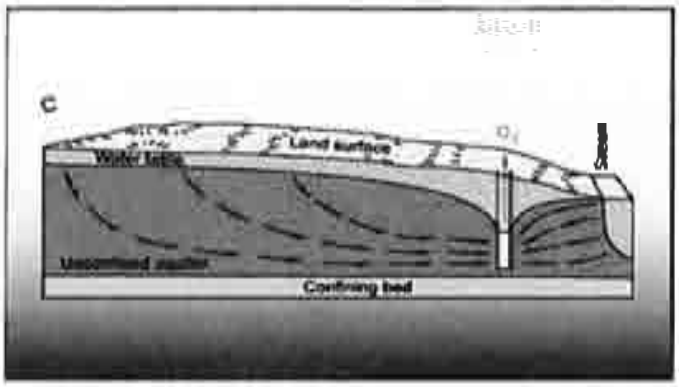

Figure 5: Drawdown of Aquifer From Pumping (Source: www.learner.org)

Monitoring and testing of groundwater in reclaimed strip coal mines indicate that groundwater is stored in and flows through large voids or conduits in spoil. However, these voids are not always connected across a mine site (Hawkins 1998 and Hawkins and Aljoe 1992). Because groundwater does not flow through coal mine spoils as it would through unconsolidated alluvium it would be very difficult to design and restore phreatic groundwater flow to stream 2003 or insure that groundwater flow to streams 2002 and 2004 would be maintained.

4.7. Human Use page 4.34 Commercial, personal and subsistence harvest: Although the reported average commercial harvest of approximately 33,000 sockeye and 33,000 coho from the Beluga and Tyonek Districts (which includes fish from the Chuitna River) may not seem large when compared to other fishing districts in Cook Inlet, these are healthy runs which can continue in perpetuity as long as the habitat is intact. To put this into perspective the sockeye harvest in these districts is almost twice the 2008 Canadian commercial harvest of 16,000 sockeye from the Frazer River (Pacific Salmon Commission 2008). The Frazer River was once one of the largest sockeye salmon producing systems in the world, but since the 1990's the runs have declined precipitously. Some Frazer River coho and steelhead stock are so depressed they have been classified as endangered. Fisheries and Oceans Canada (2008) attributes the decline to watershed development including mining, agriculture, logging and global warming. Even though the Canadian government has attempted to arrest the decline by spending millions of dollars on the same types of habitat restoration and enhancement proposed for the Chuitna Coal Mine, these efforts have been largely unsuccessful.

Summary:

1. The report does not explain how one year of CPUE data from minnow trapping can be used to determine the impact of strip mining on stream 2003 or how it will be used in stream restoration. The assumptions, limitations and problems with using minnow traps are not presented or discussed in the report. CPUE does not provide an estimate of the population size of each species of fish which is needed to assess impacts and restoration success. Juvenile salmon populations and CPUE may change substantially from year to year and from stream to stream. For

example, over a 17-year period the estimated juvenile coho population in the Kenai River ranged from 400,000 to 1,200,000, a change of over 300% (Massengill 2009).

2. Aerial and foot adult escapement surveys do not provide accurate estimates of escapement levels, especially in small streams. A weir or sonar counter is needed to enumerate adult escapement into the Chuitna River and into streams 2002-2004. Because of substantial annual variations in run size, a minimum of 5 to ten years of continuous escapement data is needed.
3. A comprehensive understanding of stream flow and groundwater flow into streams 2002-2004 is necessary to determine the instream flow needs of fish and other aquatic life and to assess impacts of strip mining and to restore watersheds impacted by mining. However, no comprehensive instream flow study and no study of the input of shallow groundwater by sub-basin have been conducted. To account for seasonal and annual fluctuations in groundwater, a minimum of 5 to 10 of years of data are needed (Rice 2009, Zhou et al 2004, and Moore 2009).

C. Chuitna Coal: 2007 Aquatic Biology Studies Program

I. Introduction

Page 1: The purpose of the 2007 Oasis aquatic biology studies program was to supplement and enhance the baseline information gathered in 2006. The purpose of the 2006 study was to determine resources and habitats in the Chuitna drainage that could be affected by the project, including streams 2002, 2003, 2004, Chuit and Three Mile Creeks and the mainstem Chuitna River .

Analysis and Comments:

2.1 Methods:

2.1.1 page 2.1 paragraph 2. The selection of minnow trapping sites based on what the researchers believed to be good habitat (e.g. good cover and low water velocities) may have biased the results. Traps should have been placed using a randomly generated sampling strategy in all types of habitat and the catch rates should have been used to identify preferred coho habitat in streams 2002-2004. Variations in water levels and water velocities at the trap sites over the sampling period would also affect the catch rate. High water levels and increased water velocities would displace coho from trap sites to slower waters. High water velocities could result in a narrower cone of dispersal, carry the scent further downstream, and make it more difficult for fish to swim upstream to the trap. Lower water levels and velocities would have the opposite effect. Catches from traps placed in shallow and faster water would have helped support the researcher's concept of good coho rearing habitat.

Minnow trapping occurred at the same sites in 2006 and , but the report did not indicate whether or not there had been any changes in the stream channel (i.e. cover, water velocities, or substrate) at those sites, which could have affected the catch rate.

2.1.1 paragraph 3 page 2.2. Why were all of the juvenile Coho from streams 2002-2004 and the Chuitna pooled for the length frequency analysis? A separate analysis could have shown if there are morphological differences (i.e. size, condition etc.) between coho stocks in these streams. Morphological differences, which may be the result of genetic isolation, have been documented in scientific studies of coho stocks in nearby tributaries of small coastal watersheds (Bailey and Irvine 1991). This is important because if observed differences were the result of genetic isolation and adaptation to the unique conditions in these streams, it would make the task of successfully restoring stream 2003 fish populations much more difficult.

Figure 2.1.1. It could be problematic that all of the minnow trapping locations were located at or below the road crossing. The effects of road construction include short term elimination of benthic life and fish population for some distance below the crossing site due to siltation and turbidity, as well as obstruction of adult and juvenile upstream migration due to poorly installed and maintained culverts, floods etc. (Barton 2006 and Moore et al 1999).

Figure 2.1.2. Why wasn't there any winter trapping in streams 2002 and 2004 to identify and characterize overwintering habitat? Suitable overwintering habitat associated with the input of groundwater is believed to be a critical factor in freshwater salmonid survival in cold climates (Fisheries and Oceans Canada 2009, Mouw 2005, and Baxter and McPhail 1999). Streams 2002 and 2004 and their fish populations are almost certainly going to be adversely affected by changes in surface and groundwater temperature and flow resulting from deep mining in their watersheds. It is possible that these streams could change from gaining streams to losing streams due to groundwater pumping to dewater the mine (USGS 2008 and Hancock 2002).

2.1.1 Page 2.6 paragraph 1. The statement that "CPUE was calculated by dividing the catch by the number of hours fished and multiplied by 24 hours for a nominalized trap catch for 24 hours" needs some clarification since the same paragraph indicates that the traps were fished for 24 hours! Further, the use of the CPUE's from these traps needs to be qualified because in areas of high population density, catches of juvenile salmonids probably reach a peak after 1-2 hours while catches in areas of lower density may continue to increase over 24 hours (Swales 2008). Because of this phenomenon, the relative densities of coho populations estimated from 24 hour trap soaks may be too low for high density areas and too high for low density areas.

3. Results and Discussion:

3.1.1.1. Fisheries Resources: page 3.1 paragraph 2. The explanation for the use of percent relative composition (i.e. percentage composition provided a way to discuss the relative numbers of species captured in minnow traps and make inferences on the

communities which reside in those streams) instead of CPUE is unclear. Why not use both? CPUE gives the reader an idea of the relative population densities by stream. A comparison of CPUE by species by year can also provide an indication of changes in population numbers and community composition (i.e. a 50% decline in CPUE might indicate a change in population numbers). CPUE is also commonly used in fisheries studies for that purpose.

Page 3.1 paragraph 3. It is generally believed that pink salmon fry in coastal streams do not feed in fresh water but migrate directly out to marine waters. In freshwater, chum salmon fry feed on insect larvae for some time before outmigration. It is not likely that they would be captured in smolt traps baited with salmon eggs. Also, age zero fry of all species and possibly some age 1 fry are small enough to swim through the mesh used in minnow traps. This means that these species and age groups may have been present but underrepresented in the catches.

Page 3.1 Coho salmon: The use of percentages of catch by species in the catch does not seem to be that useful for purposes of environmental impacts assessment, permitting and mitigation. The actual number of each species captured would have been more useful. For example, 88% of a catch of a million fish (880,000) is a lot different than 88% of a thousand fish (888). It is unclear whether the percentage figures (63-88%of the fish population) are for coho fry alone or if they include spawning adults.

Page 3.3 Paragraph 3 Sticklebacks: The large increase in stickleback numbers in the Chuitna and Threemile Creek could be explained by the fact that there are anadromous stocks of sticklebacks in Cook Inlet and they enter streams in large numbers in some years.

3.1.3.2 Comparing Rosgen Stream Types Within Study Streams: Page 3.15 paragraph 2: There are many different types of Rosgen B, E and C channels based in part on slope. The statement that "Rosgen type B channels typically exhibit the highest gradient and current velocities compared to C and E channel types" is accurate only for B1a-B6a channel types (Rosgen 1994 and 1996). B1-B6 and B1c-B6c have the same slopes as Type E channels and C 1b-C6b channels. The authors should specify what types of B channels they are reporting on, especially if this information is going to be used in developing mitigation and reconstructing stream channels.

Page 3.16, paragraph 1: The speculation that Rosgen type C channels supported the highest concentrations of spawning adult coho and also had the highest CPUE of all channel types, and that this might be related to nutrients provided by adult salmon eggs and carcasses is important. It is important because mining will eliminate these stream channels and the anadromous fish stocks which spawn in them.

There is a large body of scientific literature showing that Pacific salmon are the major vehicle transporting marine nutrients across ecosystem boundaries from marine to freshwater and terrestrial ecosystems. Nutrients from salmon eggs and carcasses play a major role in the productivity of both freshwater and riparian ecosystems and in perpetuating future salmon runs. Most fisheries scientists and fisheries managers have concluded that stream ecosystem health benefits from having the largest number of spawning salmon possible (WDFW 1997), which in turn produces a large number of carcasses. The eggs and carcasses from these spawning salmon provide an essential source of food for rearing salmon and other fishes which concentrate in these areas. Nutrients from decaying carcasses also provide food and nutrients for insects such as chironomids which are the major food source for salmonids during the rest of the growing season. Bilby et al (1996 and 1998) found that benthic algae, invertebrates and fish in salmon streams were significantly enriched with both marine carbon and nitrogen. The average contribution of marine nitrogen ranged from 11% for invertebrate predators to 31% for juvenile Coho. The highest percentage of marine nitrogen was 46% for adult cutthroat trout and 61% for age 1 plus steelhead. The same researchers also found that the growth rate of juvenile coho doubled after adults spawned in the stream, where as in a nearby stream without spawning salmon juvenile steelhead showed no increase in growth rate during the same time period. This phenomenon is so important that fisheries scientists recommend that escapement goals should be designed to produce "nutrient capital" within watershed that will help support the next generation of fish.

During mining, salmon will be excluded from the middle and upper portions of Stream 2003 where most spawning and rearing occurs for a long period of time. The "nutrient capital" built up over hundreds of years would be lost when the drainage is destroyed. Diminishing or eliminating salmon production (i.e. eggs and carcasses) from a stream due to natural or anthropomorphic causes such as strip mining may be self-perpetuating. Without necessary nutrients from salmon eggs and carcasses remaining downstream, stream 2003 stocks are likely to decline further. A restored stream drainage without nutrients from salmon carcasses is not likely to be very productive (Bilby et al 1996 and Larkin and Slaney 1997). The concurrent loss of most of the wetlands, which are the other major source of stream nutrients in the stream 2003 drainage, will further reduce stream productivity (Hood et al 2008, Meyer et al 2003, and Nagorski et al 2007).

Significant loss of stream productivity from premining conditions has been documented in studies of streams in reclaimed stripmines. Matter and Ney (1981) found that 5 to 7 years after reclamation, "benthic invertebrate and fish populations were significantly lower in abundance in the reclaimed mine streams than in the reference stream and showed less taxonomic richness and stability. Biota of reclaimed streams were similar in these respects to the biota of unreclaimed mine streams."

Page 3.16 Paragraph 3. The authors suggest that with some distinctions, data from streams 2004 and 2002 may be similar to stream 2003, and as "mine development progresses impacting stream 2003, streams 2002 and 2004 can be used for comparative purposes to stream 2003." With enough years of data to determine fluctuations in the abundance and composition of fish populations with statistical confidence, streams 2002

and 2004 might provide some general comparison with stream 2003. However, it seems likely that stream 2004 and possibly stream 2002 may be also impacted by changes in surface and groundwater flow and water quality from mining in their watersheds. If so, these impacts would negate any objective comparisons.

3.1.3.3 Longitudinal Comparison of CPUE between Streams:

3.1.3.4 Study Streams Coho Salmon: page 3.-19 paragraph 3. The admonition that "just because C channel produces higher CPUE than other stream channel types it should not suggest that the length of stream 2003 be rebuilt as a C channel in reclamation planning and construction" is good advise. Reconstructing a stream channel to approximate any of the physical characteristics of Rosgen stream types (i.e. entrenchment ratio, width depth ratio, sinuosity, slope etc.) does not provide any assurance that the new waterway would provide the key characteristics of productive salmonid habitat, such as hyporheic flows, groundwater upwelling, nutrient input from adult salmon, and the 2000 acres of wetlands in this drainage. Although, there are hundreds of thousands of streams in Alaska with Rosgen B, C, and E channels, less than 20,000 are have been found to support spawning, rearing or migrating salmonids and a fraction of those have the unique conditions which provide spawning habitat for coho, or Chinook salmon.

Although there are examples of stream channels that have been reconstructed, rehabilitated or relocated in Alaska, all of these projects have entailed restoration of stream banks, movement of short sections stream channels within an existing flood plain or relocation of a stream to an old stream channel.(ie Moose Creek). I was also unable to find any pre- and post-construction quantitative scientific studies documenting the biological and physical effects of these projects on fish populations.

3.1.3.1.. page 3-36, paragraph 2. The statement that "juvenile Chinook may emerge from the gravel to migrate to sea at age zero: therefore they may not be present during the winter during our trapping" is not consistent with the increased winter Chinook catch rates shown in Table 3.1-11. A large percentage of age zero Chinook do smolt in some systems, but all studies to date indicate that Cook Inlet Chinook smolt at age 1 (Hasbrouck 2009, Kerkvliet 2009). It is important to establish whether the Chuitna Chinook smolt out migration at age 1 or age 0 for impact assessment, permitting and mitigation purposes.

The authors could be correct that pink and chum fry were not free swimming during the time the minnow traps were in the streams; however, in my experience pink, sockeye and chum salmon fry are usually not caught in minnow traps. Often they are so small that they can swim through the mesh in the minnow traps, and they may not be attracted to the salmon egg bait.

Table 3.1-12 page 3-37. There appears to be something wrong with the average length column for table 3.1.12. Aside from the fact that the column has no unit of measurement, e.g. millimeters or centimeters, the average lengths given do not seem to be correct. For

example, it seems unlikely that a juvenile coho salmon would ever be as small as 8mm or .323 inches, or that a fish that small could be captured.

3.1.3.12 Electrofishing. Was electro fishing conducted at the same time as trapping so that the same fish populations were sampled? If not, it would not be safe to conclude that the minnow traps were capturing all of the species and age groups present.

3.1.4.1 Spawning surveys page 3-40. Foot and aerial surveys usually significantly underestimate the number of spawning salmon present in a stream (Jones et al 1998). To accurately enumerate the number of salmon spawning in Stream 2003, a weir or sonar counter should have been installed and operated for 5 or more years. Accurate weir or sonar counts of adult salmon from the mainstem Chuitna and streams 2002 and 2004 are also needed.

Page 3-81, paragraph 3. This paragraph states that the majority of the data necessary to reconstruct/design stream 2003 has been collected and all that remains is to establish exact channel positions (plan form) and compile streambed elevations through the mine. It further states that "Data collected to date has been sufficient to characterize existing reference conditions, and to provide evidence of restoration feasibility." Oasis believes that they have almost all of the data and the expertise to reestablish a functioning salmon spawning stream after all of the existing vegetation has been removed, the entire upper stream 2003 watershed is excavated, the underlying substrate excavated to a considerable depth, and the surface hydrology and the existing shallow aquifer completely disrupted.

I disagree that PacRim has collected all the data necessary to reclaim stream 2003 and its drainage. For example, no data is provided on present shallow groundwater flow or the techniques that could be used to successfully restore essential groundwater flow to a reconstructed stream 2003. PacRim has not studied or provided any data on the role of marine nutrients from salmon carcasses or nutrients from wetlands in the productivity of the stream 2003 drainage. There is no explanation of how they would restore the thousands of acres of wetlands that would be destroyed by mining. Most importantly, they haven't provided any examples from the scientific literature of successful restoration of the type and scale they are proposing.

Although I have over 25 years of experience in analyzing and permitting projects affecting fish habitat, including large and small mines in Alaska, I am not aware of any instance where a salmon spawning and rearing drainage has been completely altered in the manner proposed for the Chuitna coal mine and then restored to its original productive state. There have been cases where small sections of previously mined stream channels have been altered to provide more productive salmon habitat and by chance has provided spawning habitat, such as the Forest Service's restoration efforts on the Resurrection River near Hope, Alaska, but no restorations of the type and scale proposed for stream 2003.

After mining the entire stream 2003 drainage and portions of stream 2002 and 2004, the drainages will consist of highly erodeable mine spoils, and water quality, including silt and turbidity, will be an issue that will affect the success of attempts to restore salmon habitat. The recovery of a stream affected by coal strip mining was monitored after mine closure by researchers from USGS (Collier 1963). The study found that mining significantly changed the chemical quality of the groundwater and surface waters, increased the sediment yield, and adversely affected the aquatic life of the stream. Even though the mined area only included 6.4% of the drainage basin, it contributed 96% of the sheet erosion. The stream affected by mining discharged 2,800 tons of sediment per square mile, whereas the control stream in an adjacent unmined drainage discharged only 49 tons per square mile during the same period. Clearing and developing land for mines increases sediment input into streams, lakes and rivers (Environmental Protection Agency, 1999, 2006 and 2008). Glancy (1973), who studied sediment loads in tributaries to Lake Tahoe, estimated that sediment yields from undeveloped areas ranged from 19 to 420 tons per square mile, whereas sediment yields from developed areas ranged from 700 to 5000 tons per square mile.

Because much or all of the vegetation in the mined areas will be removed, it seems likely that stream and groundwater temperatures will increase in summer and decrease in winter to the detriment of spawning and rearing salmonids (Beschta and Taylor 2007, Conjack 1996, Feller 1982, and Curry et al 2002). There is no mention of this issue in the report, or of how restoration efforts will address detrimental changes in water temperature until mature vegetation is reestablished.

3.8.1 Landscape scale investigations page 3-102 paragraph 2. Upwelling groundwater and adequate instream flow is essential to successful salmonid spawning and juvenile overwintering survival in Southcentral Alaska (Alfredsen and Tesaker 2002, Baxter and Hauer, 2000, Malcolm et al 2004 and Mouw 2005). In section 3-81, Oasis purports to have almost all of the data necessary to restore Stream 2003 to productive salmon habitat. Yet, in this paragraph Oasis indicates that they don't have the detailed information on groundwater location, flow rate, temperature etc necessary to restore stream 2003: "The predominantly glacial alluvium lacked distinct surface nick points in the geological stratigraphy to force hydraulic pressure upward at longitudinal breaks." Also: "Landscape features were not available in steam 2003 that would serve as predictors of hydraulic exchange, and spatial heterogeneity, suggested areas of local hydraulic exchange."

Landscape Scale Investigations Page 3-106, paragraph 2. Although most of the findings in paragraph 2 are consistent with other studies of hyporheic exchange in salmonid spawning areas, the finding that "most adult salmon selected downwelling zones for spawning" may not tell the entire story about the dynamics of these spawning areas. Other researchers have found that in areas where freezing can destroy salmon embryos, salmon seem to favor sites with upwelling groundwater, which will be warmer in winter (Quinn 2005). This behavior is documented in sockeye, coho, Chinook and chum salmon in Alaska (Quinn 2005 and USFWS 2009). A comprehensive study of redd site selection by bull trout (Salvelinus confluentus) found that they selected stream reaches for

spawning that were strongly influenced by upwelling groundwater but located redds in transitional bed forms that processed strong localized down welling and high intergravel flows (Baxter and Hauer 2000). Similarly Malcolm et al (2004) found that Atlantic salmon (Salmo salar) selected areas of complex groundwater-surface water interactions and that during critical freezing or low flow conditions hyporheic conditions were dominated by groundwater, whereas surface water dominated during high flow conditions. Additional research is needed on the influence of groundwater on redd site selection for all species of salmonids in streams 2002-2004.

Page 3-113 paragraph 1 . Oasis makes the point that there are significant differences between streams 2002, 2004 and 2003, and that these differences should be taken into consideration when comparing these streams to stream 2003 during mining. This conclusion is correct. The physical and biological differences between these streams and the possibility that these streams may also be affected by mining raises serious questions about using these streams as controls or templates for stream 2003 restoration. The type of comparison which might be appropriate is this: If there were normal populations of adult and juvenile salmon in streams 2002 and 2004 and few or no salmon in the restored sections of stream 2003, it would likely indicate that the restoration efforts were not successful. However, if there were 10 or more years of comprehensive pre-mining physical and biological baseline for stream 2003 to be used as a model for reconstruction, such a comparison would not be necessary.

Page 3-113: Velocity paragraph 2; Velocity: Oasis reports that "water velocity and currents could have significant effect on (minnow trap) catch rates (Bjorn and Reiser, 1991) and both influence habitat use by juvenile fish." This statement is correct. However, catch rates can also be affected by water temperature (too high or low), turbidity, and time of day. These factors were not reported on or discussed in the report (Bjorn and Reiser, 1991).

Oasis measured water velocities at each trap site and found that "Velocities ranged from 2.0 feet per second to 0 feet per second with 50% of the sets having velocities less than .5fps and 90% of the velocities less than 1fps." They reported that "juvenile trout, salmon and char (40 to 180 mm in length) usually occupy sites with velocities up to 1.3 fps (Bjorn and Reiser 1991) and juvenile coho salmon prefer pools with velocities less than .7 fps (Bisson et al 1988)." They concluded that the preferred velocities are close to or below over 90% of typical trap site velocities, therefore velocity was not a limiting factor for catching fish at individual trap sets."

I checked Table 4.10 in Bjorn and Reiser (1991) for depths and velocities at sites used by salmonids in streams. The velocities reported for age 0 coho salmon in natural habitat ranged from .16 fps to .98 fps and for age 1 coho from .59 to .98 fps. This is lower than the range of preferred velocities reported by Oasis and would indicate that the percentage of time that water velocities at the trap sites exceeded preferred water velocities was greater than estimated. Preferred velocities for age 0 Chinook salmon are similar to juvenile coho salmon (Bjorn and Reiser 1991).Water velocities well above preferred velocities would likely reduce catch rates at times, particularly at high flows when

turbidity would also increase. At high flows, juvenile salmonids and other fishes would move to areas with lower velocities. Fluctuating catch rates along with the other limitations of minnow traps as a scientific sampling tool add uncertainty to the data and limit the value of the minnow trap data to compare catch rates over time and between streams.

Page 3-115. Depth. I am not sure that it is scientifically sound to conclude that "Given that 90% of trap sites were in water less than 3 feet deep, combined with the belief that preferred water depths will be readily subjugated to access to food, and security to predators, and that traps were set in a minimum depth so that the trap entrances were submerged, it is not likely that water depth limited or biased catch rates."

I could not find any source in the literature cited section for the citation that "Juvenile coho salmon show a maximum preference for depths of 3.3 feet (Beecher et al 2002)." It is also not clear from the Oasis report if this means that juvenile Coho salmon prefer depths of 3.3 feet or that this is the maximum depth that they are found at. The article cited in the North American Journal of Fisheries Management actually stated that "Juvenile coho showed maximum preference for depths of 76-100 cm (2.49 feet-3.3 feet) and velocities of 3-6cm/s (.098- .196 f/s)" (Beecher et al 2002). Velocities of .098 .196 f/s indicate very slow moving water that would typically be found in pools or near stream banks! Other researchers have reported that at water velocities of 0.30 to 0.98 f/s, juvenile coho salmon preferred shallower water (i.e. 0.98-2.2 feet) (Nickelson and Reisenbichler 1977). It appears that there is a relationship between coho presence, water depth and water velocities that is not accounted for in the Oasis conclusions. If coho moved from a trap site in response to fluctuating water levels or velocities, which occurred a number of times over the study, it would affect catch rates. Because Oasis apparently did not constantly monitor water levels or velocities at trap sites there is no way to compare catch rates to these parameters to see what effect they may have had.

The statement that juvenile Chinook salmon have been found in depths ranging from .5 to 4.5 feet (Bjorn and Reiser, 1991) is not meaningful by itself. I have captured juvenile Chinook in dip nets in a few inches of water and in inclined plane traps in 8 feet of water. This does not mean that those depths are preferred habitat for juvenile Chinook salmon, or that minnow traps placed in those areas would provide an unbiased index of abundance. The 5 studies cited in Table 4.10 in Bjorn and Reiser (1991) indicate that most age 0 Chinook fry were found in water depths of .5 feet to 2.0 feet of water at water velocities of 0.20 to 0.98 f/s. It seems likely that when water depths or velocities exceeded those depths and velocities, catches would be reduced to some degree, affecting CPUE.

Another factor that was not discussed, but that could have negatively influenced juvenile Chinook catches, is that juvenile coho could have displaced juvenile Chinook from areas with minnow traps because the traps contained a favored food source (i.e. salmon eggs). Coho fry are aggressive and territorial soon after emergence, and they establish intraspecific dominance hierarchies (Mason and Chapman (1965). Where coho and

Chinook fry occurred together in streams, the coho were socially dominant, defending territory accessible to incoming food (Stein et al 1972).

Page 3-116. Temperature. The Oasis data does not appear to support the conclusion that "since temperatures show a consistent pattern across drainages, temperature is not considered to significantly bias CPUE between drainage. *Rather catch rates changed throughout the season, equally between drainages.*" A comparison of Figures 3.1.5-3.1.5 coho salmon CPUE and the accompanying tables by month and location for streams 2002-2004 indicates that CPUE did not change "equally" by stream over the season. The CPUE curves between streams and within streams between May and September appear to be significantly different. For example, as depicted in Figure 3.9.4, catch rates in streams 2002 and 2004 declined in September as water temperatures declined. However, 2007 catch rates for all sections of Stream 2003 actually increased significantly in September! This suggests that either the temperature measurements are incorrect or catch rates were influenced by some other factor than temperature.

Another factor that was not investigated but could have reduced catches of Chinook and rainbow trout is that juvenile salmonids that live in streams are sometimes nocturnal and may not feed during the day (Bradford and Higgins 2001). Little is known about diel activity of salmonid movement patterns in Alaska, however, researchers in British Colombia found nocturnal diel activity patterns in Chinook salmon and rainbow trout in areas of high flow in the Bridge River. Parr and older fish were more nocturnal in summer than fry, and all fish were nocturnal during the winter. They concluded that habitat-driven variations in activity patterns will likely affect the processes that regulate these populations and could make the prediction of the effects of ecosystem manipulation such as changes in flow very difficult.

Summary:

The problems and limitations of using minnow traps and other types of passive fishing gear in scientific studies are well known (Hubert 1996). Clear selectivity leads to misrepresentation of certain types (e.g., particular size or species) of fish relative to their actual abundance in the environment (Hubert 1996 and Breen and Ruetz 2006). All passive fishing devises are selective for certain species, sizes or sexes of animals.

There is so much variation in CPUE within and between streams 2002-2004, and between years, that it is difficult to imagine how this data could be used to monitor mining-related changes in stream 2003. If a consistent pattern exists, it would take many more years of data collection to determine whether such a pattern exists and what it is.

It is not clear how the emphasis on CPUE and percent relative composition fits into the regulatory requirements for avoidance, mitigation and replacement. An estimate of the actual numbers of adult and juvenile salmonids produced by streams 2002-2004 from sonar and weir counts would be more useful to assessing impacts and developing mitigation and monitoring. The data from minnow trapping is not likely to produce a population estimate with any degree of confidence. Because of the natural variation in

adult and juvenile salmonid populations in Cook Inlet, 5 to 10 years of sonar or weir data would be necessary to provide estimates with any degree of confidence.

It appears that the objective of collecting morphological stream data may be to meet permitting requirements by replacing the stream miles of Rosgen type B, C,and E that will be destroyed with reconstructed habitat that meets the characteristics of B, C, and E habitat. It is not clear how merely duplicating the general physical characteristics of stream 2003 can be considered successful restoration. For example, this is no discussion of how the flow of hyporheic groundwater to stream 2003 would be restored. One of the most critical factors in stream restoration appears to be the input of groundwater to the hyporheic zone. Ground water contributes cooler water to the instream flow during the summer , and warmer water which comprises the total flow during the winter months. This flow, which is near the mean annual air temperature, controls the development rate of salmon eggs and allows juvenile salmonids to successfully overwinter.

D. Wetland Functional Assessment (HDR Alaska, Inc., March 5, 2008);

HDR's report is a supplement to the Baseline Report for Vegetation and Wetlands (HDR Alaska 2007b). The baseline report detailed how wetlands were identified and mapped. This report addresses the assessment of functions performed by wetlands identified in the baseline study.

Analysis and Comments:

This report provides important information on the functions provided by the 3,921 acres of wetlands that will be destroyed by strip mining. HDR lists 6 functions those wetlands provide, including: wildlife habitat, stream flow moderation, shoreline stabilization, groundwater discharge, groundwater recharge, and carbon export food/chain support. Two additional wetlands functions that are not identified in the assessment are water quality maintenance and fish habitat. Wetlands filter out pollutants and trap silt and sediment that would otherwise enter fisheries habitat. This becomes increasingly important in disturbed areas such as a strip mine. Wetlands that are connected to streams can provide important fish habitat, particularly for juvenile coho salmon.

Results: Page 13. HDR estimates that there are 4,000 acres of wetlands in the Chuitna Coal Mine Mapping area, or 43% of the mine area. They estimate that the acreage of wetlands within the mine area providing the following important functions are: Stream flow moderation, 340 acres; groundwater discharge, 1,646 acres; groundwater recharge, 24 acres; carbon export/food chain support, 2,120 acres; and total functional wetlands acreage by component, 3,023 acres (HDR 2008).

According to HDR, the total acreage of wetlands providing groundwater discharge in the project area is 2,171, and the total acreage of wetlands providing carbon export /food chain support is 4,174.

Page 22: Wetlands to be destroyed by strip mining currently provide the following functions in the Chuitna Coal Mine Area: 9% of wetlands providing stream flow moderation, ; 1646 acres or 42% of the wetlands providing groundwater discharge,; 1% of the wetlands providing groundwater recharge,; and: 56.5 % of the wetlands providing carbon export/ food chain support to streams,2002-2004,.

Page 24: Total functional wetlands acreage in the Chuitna Coal Mine Area: 3,023 acres, or 77% of the total of 3,921 acres of wetlands in the mine area, provide essential functions for streams 2002, 2003, and 2004

Summary:

The productivity of a salmon stream is based on marine derived nutrients (MDN) from salmon carcasses and the flow of organic matter, nutrients, and the consistent flow of high quality groundwater and surface water from its drainage basin (Piccolo et al 2009, Mathisen et al 1998, and Schlosser 1991). Wetlands have been identified as a major terrestrial contributor of organic matter and nutrients to salmon streams (Hood et al 2008, Pess et al 2002, and Nagorski et al 2007). All of the wetlands, which currently comprise 43% of the proposed mine area and currently provide groundwater discharge, groundwater recharge, and carbon export/food chain support to streams 2002, 2003, and 2004, would be destroyed by mining. There is no plan to replace these wetlands, and it may not be technically or economically feasible to construct thousands of acres of replacement wetlands. At the same time, both the reservoir of MDN within the mine area and the input of MND from salmon carcasses will be lost when the stream 2003 channel and riparian area is mined. Even if stream 2003 could be reconstructed, the loss of both of the major sources of stream productivity (i.e. marine derived nutrients and wetlands) would make it very difficult to restore it to its former level of productivity. The loss of a portion of the wetlands in the streams 2002 and 2004 drainages will also affect the productivity of these systems.

E. Chuitna Coal Project: Oasis Environmental Inc. Winter Freshwater Fish Habitat Baseline Report, January 2009.

I..Introduction:Oasis'stated objective for the *Oasis Environmental Inc. Winter Freshwater Fish Habitat Baseline Report, January 2009*, was to determine the location, longitudinal movement and habitat use of overwintering fish (primarily juvenile coho salmon and small resident Dolly Varden) within randomly selected segments of stream 2003 within the proposed Chuitna Coal Mine project area during the winter of 2007/2008. The purpose was to satisfy permitting requirements and to obtain an understanding of winter habitat use to assess and mitigate habitat loss and biological impacts of the proposed project.

Analysis and Comments:

II. Methods:

Section 2.3.1, Page 16: Oasis reports that the estimated tagging mortality for juvenile coho salmon was 4% and 0.7% for Dolly Varden. Given the description in Section 2.3.2 of the difficulties encountered in capturing, holding and tagging fish under very adverse conditions (*e.g.* fish freezing to the tagging boards), it is surprising the mortality was so low. No information was provided in the report on the number of fish that died before being released. Although the process of handling and injection of foreign material into very small fish always entails a risk of infection or physical injury, no estimate of long term mortality after release into stream 2003 was provided. A Peterson mark and recapture population estimate is based on the ratio of the total number of tagged fish released to the number of tagged fish recaptured when the population is sampled. Increased post-release mortality of tagged fish would result in an overestimate of population size.

Section 2.3.1 Page 15. It would be very useful to be able to compare the total number of fish captured during both the tagging and recapture operations. Table 4.1 indicates that a total of 2,198 coho were recaptured, of which 82 were tagged. The results section indicates that a total of 2,032 Dolly Varden were captured and 125 were tagged, but no similar number was provided for coho. It seems unlikely that all coho captured were tagged.

Section 2.3.3. Page 20. Although underwater video recording has the potential to provide some information on the presence and absence of Dolly Varden and coho in riffles and fast run habitat under the difficult conditions the researchers encountered, its use raises some questions. The disturbance from drilling holes in the ice and artificial light seems likely to displace overwintering fish from the area of disturbance. In the 1970's, I was involved in an extensive research project which involved the use of underwater video recording. My experience was that fish rapidly left the area when the camera and light system was introduced. Other researchers have reported that the lighting needed for underwater video cameras caused similar avoidance problems (Rand and Logerwell, 2009). Additionally, while bait can be used to attract fish from downstream to the area of the video camera, there is no way to determine if the fish are coming from shallow riffle habitat rather than deep pools or other habitats downstream.

III. Results:

Page 26: The statement that "variations in the number of fish (tagged per site) generally reflected the abundance of fish that were present in each particular stream segment at the time of trapping" is unclear. Does this mean that Oasis had some way of determining the total number of fish in each stream segment, which seems unlikely; or does it mean that they were not able to capture 200 fish in all sections because of low catch rates in some sections? This should be clarified

Section 3.2 Winter Movement. Page 27. The finding that tagged overwintering Dolly Varden and Coho tended to remain in the same stream sections during the winter is useful. However, there is a possibility that untagged fish from the mainstem Chuitna or other stream sections could have moved into Stream 2003 after tagging ended on October 27. The recapture of tagged fish would not have documented these movements. It is

possible that some previous study addressed this issue, but this report did not exclude that possibility.

Section 3.3 Underwater Video Recording Page 28: See previous comments on Section 2.3.3. The report indicates that video recordings were also made in streams 2002 and 2004. No reason was given for looking at these streams and it isn't clear whether or not fish were observed.

IV. Discussion:

Page 31 paragraph 1: I agree that information on the behavior and winter habitat preferences of stream fishes in Alaska is very limited. Because juvenile fish of the same species in Alaska are likely adapted genetically to very severe winter conditions (thick ice, very low temperatures, low oxygen levels, longer winters, different stream flow regimes etc.), it is not clear how much information from southern populations, including British Colombia, is applicable to stream 2003 without verification through independent research (Olsen et al 2003). Because detailed information on the life history and winter habitat preferences of Chuitna drainages fishes is essential to assessment of impacts, potential mitigation of impacts and successful restoration of stream 2003, a great deal of additional research in this area is needed (see conclusions section).

Page 31 paragraph 2: The explanation given for why overwintering juvenile coho salmon in stream 2002 were not captured in large numbers in beaver ponds in the stream 2003 drainage, whereas in more southerly coho populations beaver dams are preferred habitat, is low oxygen levels. This seems plausible. However, the study cited (Ruggerone 2000), actually found that coho fry were present in Black Lake, Alaska, which had very low oxygen and water temperature levels. Ruggerone also found that coho fry could tolerate much lower oxygen levels than sockeye fry. Oasis did not report on either winter oxygen levels or water temperatures in beaver ponds, which would have helped in the interpretation of the report.

Because information on salmonid habitat and behavior from more southerly climatic regimes and streams may not accurately represent what is going on in stream 2003, additional research is needed to determine the fine scale behavioral adaptations of overwintering juvenile coho and Dolly Varden in response to the extreme winter conditions present in the Chuitna River drainage. This information is essential to meeting the objective of avoiding or mitigating habitat loss. For example, do these fishes insinuate themselves into the interstices between the rocks where there are warmer hyporehic flows to avoid freezing? Do they seek groundwater seeps or areas of upwelling phreatic groundwater for the same reason?

Paragraph 4, page 31: The statement that "There was no change in the mean length of fishes captured throughout the winter (Table 4.1)" is not consistent with the information provided in Table 4.1. Table 4.1 indicates that the mean total length MTL of all tagged Dolly Varden was 110.0 ml, the MTL of recaptured tagged Dolly Varden was 103.0mm,

and the MTL of all recaptured Dolly Varden was 105.20mm. The mean length of tagged and untagged Dolly Varden measured in the recapture program was 5 to 7mm shorter than the mean length at the time of a tagging. The same is true for juvenile coho. The mean total length MTL of all tagged coho was 72.90ml, the MTL of recaptured tagged coho was 74.70 mm, and the MTL of all recaptured coho was 73.10 mm. Tagged and untagged coho captured in the recapture program were slightly larger than the MTL at tagging. Whether these differences in MTL are significant would depend on a statistical analysis of the two sets of data.

Paragraph 1 page 32. In simple terms, juvenile coho and Dolly Varden migrate into or remain in stream 2003 because the conditions there optimize their chances for survival during the winter. For the purpose of permitting a mine, it is important to know if other tributaries of the Chuitna and the mainstem Chuitna provide similar overwintering habitat. This is important because it appears that mining will destroy a substantial portion of stream 2003 and its aquatic resources, and mining may also adversely affect both the hydrology and water quality of streams 2002 and 2004 and diminish or displace the fish and other aquatic resources using these systems.

Paragraphs 1-3 page 33. The reasons given for lower CPUE in the winter are reasonable, i.e. coho and Dolly Varden are cold-blooded animals, and as water temperatures decline so do their body functions, including digestion and feeding. This makes the bait in traps less attractive and lowers catch rates. An alternate explanation which seems less likely, but has not been excluded, is that some fish leave the system in the fall and early winter to escape declining water levels and temperatures and migrate back in the spring.

IV. Conclusions: The study provided some qualitative information on the 3 objectives but only a limited amount of quantitative information because of the techniques used. Baited minnow traps attract and hold small fish which feed on salmon eggs from some unknown distance downstream. Large fish, age 0 and age 1 fish which are small enough to swim through the screen covering the traps, and species that are not attracted to salmon eggs are not captured. Because the coal mine would destroy a significant portion of overwintering fish and fish habitat in stream 2003 and its tributaries, information on the numbers of overwintering coho salmon, Chinook, rainbow trout and Dolly Varden using this system in relation to the remainder of the Chuitna drainage and their overwintering requirements is important. However, this study did not compare numbers of overwintering fish in stream 2003 with these other drainages. I also searched the scientific literature but was unable to find references to successful substitution of artificial overwintering habitat for natural overwintering habitat in streams that had been destroyed.

Summary:

The report provides limited but useful qualitative information on winter habitat preferences and longitudinal movements of Dolly Varden and juvenile coho tagged before October 27. Because fish were only color marked and no fish were tagged after October 27 there is no way to determine from this report if juvenile coho or Dolly Varden

moved into or out of stream 2003 after that date. Because changes in stream conditions (flow, floods, water temperature, ice formation, snow depth, location of pools and riffles), and numbers of overwintering fish could change substantially from year to year, more than one year study is necessary to meet permitting requirements. Because the proposed Chuitna Coal Mine will destroy stream 2003 and all of its aquatic life, additional studies on overwintering behavior, winter habitat preferences , instream flow, groundwater flow, and genetics are necessary to meet permitting and mitigation requirements.

F. Chuitna Coal: Movement and abundance of fish in the Chuitna River drainage, Alaska, May though September 2008, LGL, 2009.

Introduction: LGL's stated objectives: (1) describe fish movements in and out of streams 2002-2004, (2) describe the effects of development on stream 2003 on production of Chinook and coho salmon smolts, (3) estimate the proportion of fish production within the Chuitna River drainage that is contributed by stream 2003, (4) describe overwintering use of stream 2003 by resident rainbow trout or Dolly Varden char, and (5) determine species composition, estimate relative abundance of juvenile Chinook and coho salmon, and establish a time series designed to detect and measure potential effects of mine development on fish production. A time series is a set of regular time-ordered observations of a quantitative characteric of an individual or collective phenomenon taken at successive and in most cases periods or points in time. What would constitue the "time series" is not defined in the context of the proposed Chuitna Coal Mine and no explanation of how it would be used to detect and measure potential effects of mine development on fish production is provided.

Analysis and Comments:

Page 1 paragraph 2: The LGL report indicates that the work was done because stream 2003 will be "developed." A description of how mining would change stream 2004 and its drainage would help to place the study into context and make it more meaningful.

The studies of fish production in streams 2002 and 2004 are important; however, because there appear to be physical and biological differences between these streams and stream 2003, it is unlikely that these streams can be used as controls in the destruction and reconstruction of stream 2003. In addition, hydraulic changes resulting from mining through the wetlands and aquifers, which provide summer base flow and essential winter flows to these streams, may also result in changes in nutrient levels and essential winter groundwater flow. Changes of these types would confound any comparison with stream 2003, and negate the use of streams 202 and 2004 as controls or reference reaches.

1.3.2 Objective 2. Page 4: LGL proposes that coho "smolt will be the life stage monitored because the production of these fish is closely linked to habitat conditions and because the relatively low interannual variability of smolt abundances (relative to adult returns and spawning escapements) increases the power to detect a difference after an impact."

Given the choices, out-migrating coho smolt may be the best life stage to monitor, but it is not clear that coho smolt outmigration would be an effective means to detect impacts from strip mining. First, the assumption that there is "relatively low interannual variability" of abundance in coho smolt out abundances" is not supported by data from streams 2002 -2004 or the scientific literature. I discussed this with several ADF&G fishery biologists and they do not feel that coho smolt production from Alaskan streams can be characterized as having a relatively low interannual variability (Hasbrouck 2009, Hayes 2009, and Massengill 2005 . LGL's assumption appears to be based on a few years of coho smolt data from the Kenai River and Cottonwood Creek. During this period, smolt production may not have fluctuated as much as adult escapement, but there was a reported 30-35% difference in smolt outmigration from the Kenai River and Cottonwood Creek from 2003 to 2004. However, from 1993-2007 estimated Kenai River coho smolt abundance ranged from 574,000-1,327,000, a three hundred percent difference. Massengill (2009). This is not relatively low interannual variability.

Smolt abundance may have been relatively stable over a 4- to 5-year timeframe in Cottonwood Creek and the Kenai River, but no data is available to support the assumption that smolt production from streams 2002-2004 is similarly stable. Flow, water temperatures, nutrient retention and other habitat factors that influence smolt production are moderated by the large lakes in the Kenai River and Cottonwood Creek systems, but the Chuitna River drainage has no lakes.

The assumption that coho smolt production is relatively stable also assumes that streams 2002-2004 are limited by the amount of available spawning and/or rearing habitat rather than the numbers of adult coho which escape to spawn. This could be correct, but no data has been provided to support this assumption.

It is also possible that there may be natural variations in freshwater smolt survival rates between streams 2002, 2003 and 2004 that could mask the effects of mining on stream 2003. No data on smolt survival by stream is provided in the report.

Habitat related declines in salmonid populations usually occur incrementally and an annual population decline of 20-25% or much more is within the range of natural variability. Population changes of this magnitude might not be recognized as mining related until the factor causing the decline was irreversible. A Washington study evaluated eight native salmonid populations to determine if rapid, sensitive detection of a reduction in abundance was possible. The study concluded that population abundance monitoring may not provide feedback sufficiently sensitive or rapid enough to implement corrective action that prevents impacts from causing harm or exceeding an acceptable level (Ham and Pearsons 1999).

1.3.3 Objective 3, page 4, Paragraph 2: LGL proposes to use habitat modeling to determine theoretical production from stream 2003 and the theoretical contribution of stream 2003 to the Chuitna River system. This may a worthwhile exercise; however, there are some problems. Habitat quality and quantity controls smolt production except in the rare instance where a system is spawning habitat limited. The PacRim studies and reports do not address this issue.

In order to determine how many years of data collection on smolt outmigration are necessary to determine the relative contribution of stream 2003 to the Chuitna River system, it is necessary to know how much variability there is in smolt production within and between streams 2002, 2003 and 2004 over time. One year's data isn't sufficient to make this determination. Smolt production is dependent on many factors, including but not limited to the number of adult spawners, habitat quality, winter survival, predation, and floods. In nearby Upper Cook Inlet streams (Little Sustina River and the Deshka River) where Chinook and coho returns have been monitored for many years, adult returns have varied by as much as 500% over a 20-year period. At times one stream may have high escapements while the other has low for reasons that can't be fully explained. Clearly, many more years of data from the Chuitna River system are needed to determine the full range of smolt abundance. For example, how would permit or mitigation decisions be affected if next year's catches found that stream 2003 produced twice as many smolt as 2002 and 2004? Apparently the current plan appears to be to collect only one year of data before the mine is permitted, and any subsequent studies conducted after that date, would be too late to be a factor in assessing the impact and permitting decisions.

1.3.4 page 5, The Oasis overwintering study did not provide any quantitative data on rainbow trout, a popular sport fishing species, although it acknowledged that rainbow trout were present and spawned in at least one tributary of stream 2003. Different sampling methods need to be used to provide information on species such as rainbows.

3.1: Fish Movement and Abundance in Tributaries, Page 9

3.1.1 Downstream movements and abundance of fish. Page 9.

Weirs: Page 10. The construction and operation of the weirs used to capture and enumerate adult and juvenile fish may have biased the data. Because of debris loads, constriction of flow, fluctuating water levels in streams and the need for constant maintenance, weir design has to be a compromise between blocking fish movements and functioning at most flow levels. LGL described the weirs used to collect data on summer and fall fish movements in streams 2002-2004 as constructed of 1.2 m by 2.4m panels and "faced with 0.64cm mesh, .95 cm mesh, or 1.27 cm mesh." Adult and juvenile upstream migrants were enumerated by video cameras in small weir breeches.

The materials the weirs were constructed of and the manner in which passage was provided for juvenile upstream migrants probably biased the data. First, even the

smallest mesh (64cm mesh) is large enough to allow age zero coho, rainbow trout, Dolly Varden and Chinook fry to pass through it and avoid the holding box or the video camera chute (Bramblett et al 2002 and Massengill 2009). As a result these small fry may not have been captured proportionate to their real numbers. Bramblett et al (2002), who used similar weirs in a study of seasonal juvenile coho, rainbow trout, and Dolly Varden movements in Southeast Alaska, reported that "Coho fry were captured at both weirs during May and June in 1996 and 1997, but most fry were able to pass through the 6.4 mm (.64 cm) mesh of the weir panels." The larger mesh sizes LCL used to construct other sections of the weirs might allow a percentage of age 1 and 2 juveniles to pass through also.

3.1.2 Upstream movement and abundance of fish. Page 12. The LGL weirs do not appear to be designed to efficiently pass or count juvenile salmonids that were migrating upstream. Because the weirs used to capture and count fish were designed to force both adult and juvenile upstream migrants to locate and pass through a small breech in the middle of the weir, juvenile upstream migrants may have been undercounted. Newly emerged salmonid fry (25-35 mm) are relatively weak swimmers and seek water velocities of less than 10cm/s (.3 fps). Larger fish (4-18 cm) usually occupy sites with velocities of no more than 40 cm/s (1.3 fps) (Bjorn and Reiser 1991). Juvenile salmonids are relatively weak swimmers and usually migrate at the margins of streams where water velocities are lower and cover is greatest.

LGL did not identify where in the weir the location of the 1.5x1.5x3 foot video breaches were located or record the velocity through these breaches in the report. Figure 6 (Photo 6) shows a weir breach located in the middle of stream 2002. Photo 7 shows water cascading (i.e. high velocity) out of the stream 2003 weir and what appears to be the video breech.

Photo 6. The downstream entrance to the underwater video chute, electronics housing, and battery bank at the weir on Stream 2002. The top of the photo is upstream.

Photo 7. The Stream 2003 ramp for adult salmon traveling upstream being inspected on September 3, 2008. Water flow is from top right to bottom left.

Figure 6 – LGL Weirs

Because the weirs both constricted the streams and created a hydraulic head, water velocities at the breaches were almost certainly increased significantly over ambient levels and over the velocities preferred by juvenile salmonids. The drop from the weir

appears to be as great as one foot and may have also impeded or blocked juvenile upstream migration. Mueller et al (2008) tested the leaping ability of juvenile coho salmon (60-135 mm). They found that 100% were able to pass an outlet drop of 0 inches, 4.7 inches 34 %, 7.7 inches 20%, and 10.2 inches 2%.

Under low flow levels very small up and downstream migrants could have passed through the weir mesh and not been counted, biasing the results.

Video system design, configuration and operation. The description of how the video system functioned is confusing. Paragraph 4 indicates that the video system was supposed to be operated 24/7 through September. Paragraph 5 states that the system was set to record video at a rate of 7 frames per second. However, the paragraph then states that the system was set to record fish passage events at 5 second intervals through motion detection, and that most debris and juvenile fishes would not trigger the system. Does this mean that juvenile fish weren't counted? Later the report states that fish counts were estimated by subsampling the first 15 minutes of each hour, except when fish counts were estimated by counting the entire hour during period of high adult fish movement.

Page 13 paragraph 4: I am not sure about the accuracy of the method used to subtract upstream migrants from downstream migrants, i.e. if 100 of a particular group or species were sighted going upstream and 50 were sighted going downstream, the upstream count was estimated to be from 50 to 100. If 100 went upstream and 50 came downstream the net should be 50 upstream. I assume that these are adults since it wasn't specified. The method is confusing since the report doesn't indicate how much time elapsed between upstream and downstream observations

In spawning areas, salmon often tend to cycle upstream and downstream. This can be a real problem if a weir is located within a spawning area. When I set up new weirs and counting towers in the Yukon River drainage, we counted salmon upstream and downstream 27/7 all season. We subtracted upstream migrants from downstream migrants to get the net. One complication was that fish spawning in the area would cycle through the weir continuously, and spawned-out fish would often swim downstream past the weir. If LGL encountered this problem, it isn't mentioned in the report.

4.0 Results page 23.

4.1.1 Tributaries: Because the weirs were not in operation or completely sealing the streams for varying periods due to high water, combined with the fact that age zero and some age 1 fish could swim through the mesh on the weirs, it is not possible to conclude with a high degree of confidence that "Fish sampling provided a complete or near complete census of downstream migrating fish."

Video detection of fish moving downstream; Page 26. LGL mentioned species identification as an issue. It is possible that there was some error in distinguishing between juvenile coho and Chinook salmon in the holding boxes, and if so there would have been a similar error in estimating composition of the fish passing through the video

chutes. Identifying juvenile Chinook and coho salmon in the field can be challenging. To see how similar juvenile coho and Chinook salmon appear, look at the pictures of juvenile coho and Chinook salmon located at www.fpc.org/bon-da/juvsalmon.html.

Page 27 paragraph 2. LGL did not explain how they distinguished small juvenile rainbow trout from other salmonids on the video tapes. LGL apportioned video coho counts based on composition of the juvenile salmonids in the holding box because it was more reliable than trying to identify juvenile salmon from video tapes under less than ideal conditions. To see how similar juvenile coho salmon and rainbow trout look refer to the pictures located at www.fpc.org/bon-da/juvsalmon.html. Age zero and likely some age 1 rainbow trout were small enough to pass through the weir mesh and an unknown but possibly significant number may not have been counted at the weir or video chute (Bramblett et al 2000). LGL cited mesh size as a reason juvenile rainbow trout were captured in the smaller mesh stream 200401 fyke net but not the larger mesh nets or weirs used in the study.

There are other indications that there may have been problems with using the video chutes as the primary method to enumerate upstream adult and juvenile migration. First, the upstream and downstream counts for adult sockeye salmon in stream 2004. Table 8 indicates that 44 adult sockeye migrated upstream and 48 migrated downstream. Because no more adult sockeye could migrate downstream than upstream, either the upstream counts or the downstream counts are incorrect. Second, the video cameras apparently did not record over the same period of time. Tables 6-8 indicate that the stream 2002 video camera operated from June 8 through September 29, but the cameras on the other two streams only recorded upstream and downstream migrants from June 29 through September 29 and September 30 respectively. I could not find any explanation of how this apparent discrepancy was addressed in the report.

4.2.2 Run timing and biological characteristics of juvenile coho salmon: page 28.

Stream 2002: page 28. There are some anomalies in the stream 2002 catches, which raises questions about the data. First, the disclosure that juvenile coho (less than 90mm) were captured as soon as the weir was installed on June 4 and peaked two days later on June 6 indicates that downstream movement began some time earlier and that a significant portion of the run may not have been counted. LGL also reported that they used partial weirs and fyke traps intermittently between May and June 4 to enumerate migrants. These methods would not capture all fish moving upstream and downstream and calls into question the accuracy of pre-weir counts and species allocations for stream 2002-2004.

Second, in stream 2002, 2,512 fish were caught in the weir and another 6,367 were observed to have moved through the video chute. In contrast, in stream 2003, 7,394 fish were captured at the weir and 366 fish were estimated to have passed through the video chute. In stream 2004, 4,085 fish were captured at the weir and 856 fish were estimated

to have passed through the video chute. No explanation is given as to why most of the fish in stream 2002 chose the video chute over the weir. Because LGL rightly considered the weir counts to be more reliable than the video estimates, and used the species composition of the weir counts to apportion the fish in the video counts between species, this discrepancy calls into question the accuracy of the stream 2002 species counts.

Page 32 paragraph 4: Chum salmon. Like other age zero fish species, including rainbow trout, chum salmon fry are relatively small and even if present in large numbers would be less likely to be captured in the weirs and large mesh fyke nets used in the study. Chum fry also feed on small insect larvae in freshwater and might not be attracted to salmon egg bait in minnow traps (Groot and Margolis 1991).

Page 32 paragraph 6. Adult coho salmon: It is surprising that only 13 adult coho were captured in the weirs through September 30 considering the relatively large number of coho spawning in the three study streams. When coho are spawned out they tend to drift downstream and pile up on any obstruction such as a weir. Were the weirs pulled before the end of spawning?

Page 35 paragraph 1: The average size of the rainbow trout sampled in stream 200401 was 35 mm. This means that they would be able to pass through the mesh of the weir panels, which were greater than 0.64 cm and the larger mesh fyke nets and would not have been sampled effectively in the mainstem of streams 2002-2004.

4.3.2 CPUE and run timing of fish groups moving upstream

Page 37 paragraph 4. The authors report that there were distinct pulses of adult coho salmon entering all three streams on September 3 and 7 as shown in appendices E, F and G. This is consistent with data from other Cook Inlet streams which indicates that upstream coho movements are triggered by high water events (Hayes 2009 and Kerkvliet 2009). Pulses also entered 2002 and 2003, and probably 2004, but the authors indicate that "*the water in stream 2004 was too high to count fish by any method.*" LGL also reported that the inability to see into the high and turbid water "may have partially accounted for the smaller number of fish seen returning to stream 2004." This makes sense, and yet Appendix G for stream 2004 contains video and video-expanded counts for September 3, 7 and the remainder of September. This discrepancy raises the question of what was the actual size of the coho run into stream 2004, as well as whether the same high water event also affected the accuracy of counts at streams 2002 and 2003. To evaluate the data it would be useful to know what the turbidity levels were and how far from the video camera fish could be seen at various turbidity levels.

4.3.5 *Visual counts during flood events, page 39:* This section indicates that due to flood events in September when the video camera wouldn't work, and when sections of the weir were removed to allow flood waters to pass, salmon passing the weir sites were counted by observers on the banks. These counts may not have been accurate. If the

water is shallow and clear, counting salmon in flood waters in a small stream during daylight hours may be feasible. During floods, however, waters are usually high and turbid. If it is raining and cloudy and light levels are low, it is difficult to see fish moving upstream in turbid waters. LGL reported that the inability to see into the high and turbid water "may have partially accounted for the smaller number of fish seen returning to stream 2004" but did not mention this as a problem for the other streams during similar high water events.

I did not find any depiction or description of hourly fish movements in the report. Salmon run upstream 24 hours per day, and upstream movements of adult Chinook and coho salmon in Cook Inlet streams are usually triggered by high water events (Hayes 2009 and Kerkvliet 2009). The report doesn't explain how fish were counted during the hours of darkness when the water was deep and turbid. During the September 9 high water event, for example, the sun rose at 0713 and set at 2039 which means that there was only about 11 hours of daylight. There was no mention of artificial lighting over the streams in the report and no evidence of it in the photos of the weirs. Without supplemental lighting over the creek, it would be extremely difficult if not impossible to accurately count fish at night in a flood.

4.4 *Abundance of coho and Chinook salmon smolt in the Chuitna River drainage, page 39.* More research is needed on the tag and recapture methods used to estimate Chuitna River coho smolt abundance. The accuracy of Peterson type population estimates depends on at least three basic assumptions: the population has to be closed (i.e. no fish die, immigrate, emigrate, or lose their tags), the fish have to be smolting and the marked individuals must distribute themselves within the marked population so they have an equal chance of being captured (Gatz, J, and J. Loar 1988). Because coho smolt are migrating out of the Chuitna River sampling area and leaving the population and other coho smolt are moving into the sampling area from tributaries without weirs, it is not a closed system. It is also not known if smolt out migrating from the different natal streams mix uniformly within the Chuitna River population. Another problem which other Cook Inlet researchers have encountered in estimating smolt numbers from tag and recovery data is uncertainty as to whether all downstream migrants are smolting or just moving (Kerkvliet 2009).

4.5.2 Temperatures, page 40 paragraph 6. The surface water temperatures reported by LGL for Chuitna River tributaries in early May, 0-1 degrees C, are within the range found to be lethal to coho, Chinook, sockeye, and chum salmon and rainbow trout (Bjorn and Reiser 1991). Mid-winter temperatures would likely be lower. To survive the winter, fish would have to move to areas warmed by groundwater (Huusko et al 2007 and Bradford et al 2001). Bradford et al (2001) investigated the overwintering behavior of juvenile stream type juvenile Chinook salmon in Croucher Creek, a small upper Yukon River tributary. They found that most of the fish that successfully overwintered spent the winter in a 700 m reach of the creek that was downstream from groundwater sources and did not experience severe icing conditions. The study concluded that small streams may be important habitats for juvenile salmon in cold climates, especially if there is a year-round source of groundwater flow that creates conditions suitable for overwintering.

Phreatic groundwater flow is clearly the critical factor in enabling juvenile salmon to escape freezing temperatures in cold climates. Locating areas of groundwater and understanding how it is propagated and enters streams 2002-2004 is critical in determining the effects of mining on these three streams and the chances for successful restoration of stream 2003.

4.3.4. Precipitation, page 42, paragraph 2. LGL found that heavy rainfall resulted in high water events and "during these high water events that our sampling equipment was most likely to have reduced effectiveness, and in some cases halted all together." Nets and weirs plug up with debris and are often carried away by high water. This is a common problem with operating weirs, traps and nets in rivers, and it is important because upstream coho and Chinook migration is often triggered by high water and cooler water temperatures associated with increased precipitation (Frazer et al 1983, Hayes 2009, Kerkvliet 2009 and Holtby et al 1984). If most coho moved upstream during the high water periods when LGL's "sampling equipment was most likely to have reduced effectiveness, and in some cases halted all together," the adult coho counts were likely too low. However, I did not find anywhere in the report a clear explanation of how high water and equipment malfunctions may have quantitatively affected adult and juvenile or how LGL compensated for lost or impaired fishing or data collection time.

The scientific literature indicates that age zero salmonid fry may be flushed downstream during high water events (Bjorn and Reiser 1991 and Branbletts et al 2002). Some of these fry could rear in the portion of the Chuitna below the weirs or in the nearshore waters of Cook Inlet and return as adults but currently would not be counted as smolts produced by streams 2002-2004. These fry would be too small to be easily seen and counted by shoreside observers in turbid waters.

5.0 Discussion:

5.1. Overview of fish species composition in tributary streams

5.1.2 Basic run timing of key fish species. Page 44 paragraph 1-2. A study of the timing of smolt migration from the Chuitna River to Cook Inlet would help put the Chuitna fish studies in context, including the population estimates.

Page 43 paragraph 1: The revelation that the rainbow fry in stream 2004 and 2004-1 were "caught with a smaller net fyke net than was used in other systems" underscores the problems with catching rainbow fry and other small salmonids in minnow traps identified in the Oasis 2007 winter study. It is possible that rainbow and age zero Chinook fry were present in larger numbers in all three systems but were not captured by the relatively large mesh in the nets, traps and weirs used by LGL.

Page 43 paragraph 3: LGL's explanation of why the estimated 2008 Chuitna coho escapement was much larger and the estimated 2008 Chinook escapement much smaller than ERT estimates from the early 1980, i.e. natural population fluctuations, seems to be the best fit. Other possibilities include errors in the 1980's data or increased harvest of

returning adults by the Cook Inlet commercial fishery. As previously discussed, escapement data from the aerial and foot surveys used to enumerate these species have a wide margin of error. Weir and sonar counts are the most reliable. Because only a couple of years of intensive scientific sampling have occurred, it is not possible to accurately estimate what the full range of adult salmon escapements to the Chuitna may be.

5.1.2: Basic run timing of key fish species. Page 44. Paragraph 3. The finding that "The bimodal run of coho into the Chuitna watershed was consistent with observations from 1982 and 1983 (ERT 1984) when coho were detected in late July and again in August" is important. It gives additional support to the likelihood that there are genetically distinct demes of coho salmon in the Chuitna River and its tributaries. There is an increasing body of scientific evidence that coho, Chinook and sockeye salmon that aggregate for breeding at spatially defined habitats have evolved into discrete populations with special genetic or physical adaptations for those habitats or demes (Stewart et al 2004, Fisheries and Oceans Canada 2009, and Brykov et al 2004). The idea that there are reproductively isolated demes in the Chuitna system is supported by recent genetic analysis which found that juvenile coho salmon stocks collected from 8 spawning locations within the Kenai River drainage are genetically subdivided (Crane et al 2009). The genetic makeup of Chinook and coho salmon stocks in the different tributaries of the Chuitna drainage is an important factor in assessing impacts and determining the feasibility of restoring stream 2003 after 20 years of mining. If a coho deme which is adapted to the unique conditions in streams 2002, 2003, or 2004 is destroyed, it is possible it can't be replaced. The genetic composition of stocks in the Chuitna River and the three streams likely to be impacted by mining should be analyzed.

Page 44. Reported differences in Coho and Chinook run timing between the 1980's and 2008 could also be the result of the observed 3- to 5-degree C increase in water temperatures in some Cook Inlet drainages (Kyle and Brabets 2001). If water temperatures in the Chuitna River drainage are increasing, this will stress fish populations adapted to colder temperature regimes and have important implications for restoration. Temperature trends in the Chuitna river drainage need to be determined as part of the environmental studies and factored into the environmental impact assessment process.

Page 45 Rainbow trout: The discovery of a large migration of age zero rainbow trout in stream 2004-01 using a smaller mesh fyke net may be indicative of a bigger problem with the 2008 study. As stated in the report, sampling using weirs and fyke nets with mesh greater than 4 mm was not optimum for capturing small, newly-hatched fish of any species, including Chinook, coho, and Dolly Varden. For example, Bramblets et al (2002) found that most salmonid fry (less than 38mm in length) "were able to pass through the 6.4 mm mesh of the weir panels." Additional systematic quantitative prepermitting sampling needs to be done in streams 2002-2004 with smaller mesh gear to determine if significant populations of these age classes of fish were overlooked in previous studies.

5.2 Juvenile coho ecology within streams2002, 2003, and 2004,

5.2.1: Life history of juvenile coho salmon:

Page 46: Paragraph 2. If the LGL weirs and fyke nets physically blocked, impeded or created water velocity barriers to upstream migration of age 0 and age 1 salmonids, it could explain why "We (LGL) did not see a reciprocal movement upstream (summer and fall) indicating that these downstream movements represented an overall redistribution in the watershed..." Bramblett et al (2002) studied in-stream coho movements in Southeastern Alaska, but with different results. Bramblett found that juvenile coho left mainstem Stancy Creek in late summer and fall and migrated upstream into tributaries, and that a large immigration peak occurred in October. In August through December 1996, 50% of juvenile coho captured in the Tye Creek weir were going upstream. In 1997, more coho were captured going downstream than upstream, indicating that some as yet unknown factors govern instream juvenile coho movements. In 1997, most juvenile steelhead captured at weirs on tributaries moved upstream from mainstem Stancy Creek into tributaries during the fall as flows increased and temperatures decreased. Bramblett et al observed that fall juvenile salmonid movements coincided with declining water temperatures and rain-induced fall freshlets common during October in both southeastern and southcentral Alaska. Similar juvenile coho movements have been documented between tributaries in the Kenai River drainage (Hasbrouck 2009). Additional study of juvenile fish movements in the Chuitna is warranted, using sampling gear more suited to small juvenile fishes and extending later in the fall.

Page 46 Paragraph 2-5. Because the question of the contribution of age zero and age 1 downstream migrants to coho salmon production in the Chuitna is important to the permitting and mitigation process, additional pre-permitting study is warranted. Miller and Sadro (2003) found that "Nearly half of each (coho) brood year moved to the estuary as subyearlings, a portion of age zero juveniles that moved downstream during spring lived in the ecotone through summer for 8 months, then most moved back upstream to winter." After these fish spent 1 to 2 years in this ecotone, their scales would show 1-2 years residence in freshwater. Because there was no sampling in the Chuit River below the rotary screw traps or after September 30, there is no way to determine what contribution these age 1 and zero out-migrants may make to total Chuitna River coho production.

Smolt Status page 47 paragraph 1: The reports states that "Prior studies in upper Cook Inlet show that adult coho salmon returns are primarily fish that migrated at age 2 (e.g., two winters spent in freshwater) with the addition of some fish that migrated at age1." This statement doesn't resolve the question of the potential contribution of the age 0 and 1 coho fry that migrated out of streams 2002-2004, because the 2009 study did not look at what those fish did after they left these streams. These fish could spend 1-2 winters in the lower Chuitna, migrate into other Cook Inlet drainages to rear, or migrate upstream after September 30 to winter in streams 2003-2004, smolt and then return as 1-2 ocean adults.

Page 48 paragraph 1. I agree with LGL's conclusion that existing information on juvenile salmon life history needs to be "augmented with studies within the mainstem river and with studies of habitat survival by age-1 coho salmon to explain the reason for emigration from the tributaries to the river by pre-smolt coho salmon." The current studies leave unanswered a lot of important questions relevant to the environmental and permitting process. (See Appendix 2.)

Page 52 paragraph 1. I agree that LGL's "estimate of smolt abundance in the Chuitna also appears low when considering likely numbers of adult fish that return to this watershed." A 32% smolt to returning adult coho survival rate seems very high. By contrast, coho smolt to adult survival rates from other Alaska coho populations have ranged from 1.1 % to 11 % (McHenry 1977, Dudiak et al 1990, and ADF&G 2007). This high survival ratio could indicate that the number of smolt produced by the streams was much higher than the 2008 estimate indicates. However, it could also mean that LGL's estimates of adult escapement are too low because of problems with the use of aerial and foot surveys to count salmon and with the operation of LGL's weirs during high water events that were discussed in previous sections. Additional research is needed to determine smolt abundance in streams 2003-2004 and the Chuitna River as well as freshwater and marine survival rates. More accurate counts of adult salmon are also needed.

It also seems likely that the lower 8 miles of the Chuitna , which was not sampled, provides coho rearing habitat and that it could support up to 10 percent of the rearing coho in the system. It is possible that a percentage of the age 1 and age 2 coho smolt produced in this section of the drainage consist of coho that migrated downstream from streams 2002-2004 as age zero or age 1 fry. Several studies indicate that coho nomad (ie age 0 downstream migrants) life history strategy includes spring /summer rearing in esturine habitats as age zero fry and an upstream migration in fall for overwintering before smolting (Koski 2009). Estuarine rearing age zero coho have been reported from a number of coastal Alaskan streams's, including streams in Cook Inlet (Koski 2009). Because they would be from the same year classes as the smolt captured in the weirs and screw traps, the percentages of smolt produced by streams 2002-2004 would not change.

5.4 Abundance of Chinook Salmon Smolts in Tributary Streams Versus the Chuitna River Watershed. Any or all of the three possible reasons given in the report (i.e. Chinook fry tend to rear in larger streams, misidentification of Chinook fry and smolt (previously discussed), and low adult escapement into the Chuitna) could explain the low number of Chinook smolt and fry captured in 2008. A number of years of good data (10-20) are necessary to make responsible science-based decisions on the dynamics of salmon stocks. In 2007, adult Chinook escapement into the Deshka River, the closest North Cook Inlet stream where Chinook escapement is monitored, was far below the five-year average. The 2008 Chinook escapement was only 7,533, down from a high of 57,939 fish in 2004; this represents a change of over 700%(ADF&G 2009). Because salmon populations experience large natural fluctuations for reasons we do not fully understand, it is

problematic to make decisions on the impacts of a mine based on just one or a few years of data.

6.0 Conclusions and Summary of Key Results

6.1 Objective 1 page 55: Describe the movements and abundance of fish moving into and out of streams 2002, 2003 and 2004. For reasons presented earlier in this review, it is not safe to conclude that "The abundance and timing of numerous fish species moving to and from streams 2002, 2003, and 2004 were described thoroughly from late spring through mid fall in 2008." For example, it was reported that juvenile coho (less than 90mm) were captured as soon as the weir was installed on June 4 and peaked two days later on June 6. This indicates that downstream movement began some time earlier and was not thoroughly documented. A significant number of adult salmon and juvenile fish could also have moved through the weirs during high water events when the weirs and video monitoring equipment was down. The mesh on the weirs and fyke nets was too large to effectively sample age zero and possibly some small age 1 salmonids because they could pass through it.

Page 57 paragraphs 1 and 2: The low numbers of adult and juvenile Chinook salmon in streams 2002-2004 in 2008 was likely the result of natural population fluctuations and cycles rather than any fundamental change in species composition. For example, the 2008 Chinook escapement into the Deshka River, another upper Cook Inlet stream, was 1/5th, and the 2007 escapement was ½, of the 5-year average from 2002-2006.

The 2007 and 2008 Deshka River coho escapements were also ¼ and 1/3, respectively, of the 5-year average from 2002-2006, indicating that both the Chinook and coho runs into the Chuitna during the same time period could have been well below average. Experience from the Deshka River indicates that at least 5 years and up to 20 years of study may be necessary to determine the full range of escapement/production from the Chuitna system.

6.2 Objective 2: Describe the effects of development on stream 2003 on production of Chinook and coho salmon smolts. The report does not describe the effects of development of the PacRim Coal Mine on stream 2003 or the process by which the data collected on smolt outmigration from streams 2002-2004 would be used to describe the effects of strip mining on stream 2003. We can assume that the effect of the mine will be to completely remove the upper 40% of the 2003 watershed as well as the fish habitat and fish stocks that currently exist. Because this area contains most of the spawning and rearing habitat, fisheries production in stream 2003 would decline significantly. Significant portions of the 2002 and 2004 watersheds would also be removed, with attendant effects on groundwater flow, water quality and very likely on fish production. This would bias any attempts to use these streams to monitor effects on stream 2003.

There are many studies which show that the large-scale watershed clearing, excavation, and development associated with strip mining adversely impact salmonid populations

through destruction of habitat, increased sedimentation (Cederholm et al 1980, Eglin and Hubert 1993, and Glancy 1973), disruption of groundwater flow (Sengupta 1993 and Garrett et al. 1998) and changes in other physical stream and water quality parameters (EPA 1999, 2006 and 2008).

Unfortunately, the scientific methodology available to estimate cumulative effects from coal mining in the Chuitna River drainage remains limited (Harvey and Railsback 2007). Although other fisheries scientists have proposed methodologies to estimate multifactor cumulative watershed effects on fish populations no plan is laid out in the LGL report (Harvey and Railsback 2007).

6.3. Objective 3. Estimate the proportion of fish produced within the Chuitna River drainage that is contributed by stream 2003. The title of this section is misleading since the only estimate provided for all of the species of fish and age groups of fish produced by stream 2003 (i.e, Chinook salmon, rainbow trout, Dolly Varden and age 0 and 1 coho salmon that did not migrate downstream) was for out-migrating fish judged to be age 1, 2 and 3 coho smolt.

LGL has provided an estimate of the proportion of coho salmon out-migrating from stream 2003 in 2008, but for a number of reasons it is not clear that this is the real number of fish that actually left these streams. There were several problems with the study that raise questions about whether the definitive estimate for coho smolt produced in stream 2003 is really 20.8% of the total. These problems include: the inability to count or accurately estimate the number of fish during high water events ; the fact that an unknown number of fish migrated downstream before the weir was installed); age zero and likely some age 1 coho fry were small enough to pass through the mesh of the weir rather than into the trap or the video chute and were therefore not counted; because most of the downstream migrants on stream 2002 passed through the video chute rather than the trap, an unknown percentage of the fish may have been misidentified as to species or misclassified as smolt because of the difficulty of identifying small, similar-looking fish from fleeting video images; and it isn't clear that all of the basic requirements for a Peterson mark and recapture population estimate (i.e. a closed population) were met. Although LGL based the estimate of pooled 80-161mm coho smolt abundance in stream 2003 on a Darroch-Peterson best estimate of 37,424 smolt in the Chuitna drainage, the 95% confidence limits for this estimate were actually a range of 33,276 to 41,572 fish (Table 14).

Page 79 Figure 3. Figures 3 and 4 indicate that during the study period, peak flows reached as high as 400-1000 cfs, and in some instances these flows lasted for days. Problems such as these resulted in the sampling gear not fishing, partially fishing, or overflowing (weirs). Figure 5 illustrates this problem and how it would have affected counts. This figure indicates that the sampling gear was not "operating normally and fish tight" roughly 40% of the time in stream 2002, 50% of the time in stream 2004, and 60% in stream 200401. This means either no data was collected during these periods or the data collected was compromised. Because high flows would have been accompanied by

increased turbidity, it would be very difficult to observe adult salmon and almost impossible to observe small fish and salmon fry. High water velocities (greater than 2-3fps) caused by high flows and constriction of the stream channel by the weir would have prevented or impeded the upstream movements of juvenile salmonids (Bjorn and Reiser 1991). The report did not adequately address this problem.

Page 127-131 Photos 1-7. These photos (Figure 6) illustrate my concern that the weirs appear to have blocked or impeded upstream fish passage particularly for small fish, and may not have effectively sampled upstream migrants. The weirs blocked the entire stream channel and created a hydraulic head and increased water velocity in the sections of the weir that were left open. At higher water levels, the problem would have been greater. Pictures 7, 14 (stream 2002), 17 (stream 2003) and 19 (stream 2004) in particular show water cascading from the weir and creating what appears to be both a velocity and a height barrier for juvenile fish. I did not see any indication in the report that LGL measured water velocities in the breaks in the weirs or the video chutes, which were supposed to allow upstream passage, to determine if the water velocities in these breaks were impeding upstream fry movements. Velocity barriers to juvenile fish passage would have reduced the numbers of juvenile salmonids that would have otherwise moved upstream without the weirs, thereby biasing the results in favor of downstream migrants.

Summary:

LGL installed and operated a weir system to monitor upstream and downstream fish movements on streams 2002-2004. Because of difficult operating conditions and problems with the weir design, which allowed some fish to pass without being counted or identified as to species and which may have blocked or impeded upstream fish migration by small fish, the counts of upstream and downstream migrants were probably low.

Juvenile salmonids that were believed to be smolting were tagged at the weirs and recaptured in rotary screw traps (RSC) at approximately mile 8 of the Chuitna River. The ratio of tagged fish to fish captured in the rotary screw traps was used to make a modified Peterson estimate of the total number of smolt produced in the Chuit drainage and in stream 2003. Because the study area may not have been a closed population, juvenile coho salmon that may be rearing in the lower 8 miles of the Chuitna River were not included in the study; for this and other reasons, the estimate of both the total number of smolt in the Chuitna and of those produced by stream 2003 may not be correct.

This study provided some new information on the movements of fish within these streams between May 8 and September 30, 2008. The report did not, however, meet its objective of "describ[ing] the effects of development on stream 2003 on production of Chinook and Coho salmon smolts." It confirmed that wintering use of stream 2003 by resident rainbow trout and Dolly Varden char occurred, but didn't provide any information on the overwintering population that remained above the weirs and didn't migrate downstream. No information was provided on potential fish movements during breakup and after September 30. Fish were clearly moving downstream prior to

installation of the weirs in the spring, and LGL postulated that fish may move upstream in October (Section 6.4 Objective 4).

It is not clear how the report established, as it claimed to, "a time series designed to detect and measure potential effects of mine development on fish production." There was no description of how the mine would affect fish production and how the data provided in this report would be used to detect changes. This would be a very important product, and since it is critical to the environmental and regulatory processes, it is important that these impacts and the proposed process for detecting them be described for agency and public review as soon as possible.

The plan to use streams 2002 and 2004 as controls to detect changes in fish production in streams 2003 is problematic because streamflow, water quality and fish production could be affected by the mining proposed in all these drainages. Shallow groundwater flow, which appears to provide critical summer and winter refugia for fish instreams 2002 and 2004 might also be affected by mining and pumping in the stream 2003 drainage.

The LGL report provides some useful information subject to the previously discussed limitations, but one year's weir and tagging data is insufficient to draw conclusions on the life history, habitat use and requirements, overwintering behavior, abundance, survival rates, species composition and movements of fish within the Chuitna River drainage. Anadromous salmon populations fluctuate substantially from year to year due to changes in the number of spawning adult as well as fresh water and marine survival rates. Freshwater survival rates are controlled by climate variability, which in turn controls critical factors such as winter streamflow and water temperatures (Lawson et al 2004). It could take 10 to 20 years of intensive study to determine how climatic changes affect fish production in the Chuitna River drainage, and an equal amount of study to determine how fish survival is affected by changes in streamflow and water temperature regimes.. There are also questions about some of the 2008 data and how the data was collected that can only be answered with improved study methods and a number of years of additional study.

G. Part D7 Fish and Wildlife Protection Plan, Chuitna Coal Project Kenai Peninsula Borough Beluga Alaska July 2007. PacRim Coal, L.P.

Introduction:Objectives of PacRim's Plan:

1. Provide options to minimize or prevent adverse impacts to freshwater fish resources;
2. Describe mitigation options and plans during operations to offset habitat disturbances; and
3. Present design-build plans in sufficient detail to demonstrate the ability to reconstruct portions of stream channel habitat to be removed during operations and provide a framework for post-reclamation monitoring to assess overall success of habitat construction.

The Fish and Wildlife Protection Plan (hereinafter referred to as the Plan) proposes to employ several types of protection measures to minimize or prevent disturbances to fish habitat to the full extent *practical*. These measures include operational procedures, construction designs, and restoration where disturbance is unavoidable. During and after operations, on-site mitigation in the form of stream reconstruction, modification, and enhancement will occur. Offsite mitigation is an option to consider offsetting habitat loss during operations but before reclamation.

The overall approach of the plan is to minimize impacts to fish habitat; mitigate for those impacts with offsite enhancement and recovery projects; reconstruct impacted channels to a natural state that provides fish habitat in a pre-mining capacity; and measure and assess the effectiveness of these activities.

Analysis and Comments:

Page 1: The Plan states that "Stream designs are prepared as design-build plans to allow for the subtleties of streambed topography that are difficult to describe in detail in a set of plans. An experienced, on-site designer during construction will be integral to this approach, and can easily incorporate these features." Although it's not entirely clear, this seems to indicate that regulators will be asked to approve a conceptual plan for reconstruction of stream 2003 and the final design would be developed during construction. Because there is no evidence that a strip-mined salmon spawning and rearing stream and its associated aquifers and watershed has ever been successfully restored and there are questions about the use of the Rosgen methodology to restore salmon streams, a detailed stream and watershed restoration plan should be required as part of the environmental review process. The restoration plan must include examples of where the proposed methodology and techniques have been successfully used in an analogous situation.

Page 3: The Plan states that "During mining operations, physical conditions will be maintained by keeping base flow and stream flow regimes as close as possible to pre-mine conditions." I could not find any indication in the Plan or in other Chuitna Coal

studies that the appropriate reaches of streams 2002-2004 have been adequately gauged or that instream flow needs in stream 2003 are known. To maintain pre-mining instream flow needs of fish and aquatic life in stream 2003, it is necessary to measure flow continuously over a long period of time (5 to 20 years) to determine what these instream flow needs are. The Department of Natural Resources requires the Alaska Department of Fish and Game to have a minimum of five years of stream gauge data to grant instream flow reservations to protect fish spawning and rearing and migration (Klein 2009 and Westphal 2009). It would seem reasonable to require at least as much data for a project that proposes to eliminate and subsequently restore fish habitat and stocks.

The Plan states that "Stream discharges, water temperature, and water quality must be suitable during at least a portion of the migration season." This statement is troubling because some life stages (i.e. eggs, fry, smolt and adult fish) are present in the streams affected by the proposed Chuitna Coal Mine at all times. For this reason stream discharges, water temperature, and water quality must be suitable at all times. Even with modern erosion and water quality protection measures in place, coal mining will likely increase surface water temperatures, silt, and turbidity and alter streamflows (Martin and Platts 1981, May et al 1997, and Ruediger and Ruediger 1999). Altering instream flow regimes in these streams is likely to result in changes in water temperature and the wetted perimeter (ie amount of available habitat) which in turn will negatively impact fish and other aquatic life.

Page 5: The proposal to capture and remove fish from the areas to be mined and release them in the unmined sections of stream 2003 below active mining is commendable but probably futile. If stream 2003 is rearing habitat limited and at carrying capacity, these fish probably wouldn't survive because there probably isn't sufficient rearing capacity in the rest of the system to support them. Eliminating or degrading a substantial part of the rearing area in stream 2003 and possibly streams 2004 and 2002 will reduce both the spawning and rearing capacity of these streams. None of the Chuitna Coal studies have addressed whether stream 2003 and the Chuitna River system is spawning or rearing habitat limited. This needs to be determined before mitigation plans are finalized or approved.

Page 6 Water Management: The Plan claims that all discharges will meet state water quality standards and NPDES limitations and that freshwater habitats downstream of the mining activity in stream 2003 and in streams 2002 and 2004 will be maintained by providing minimum baseline flows in each stream. These claims are easy to make in a permit application but very hard to achieve in practice. For example, similar statements were made in the Red Dog Mine and Kensington Mine EISs and permit applications, but both of these mines have violated state water quality standards. Teck Cominco has been cited by the EPA for hundreds of water quality violations since the Red Dog Mine began operating in 1989 (ADN 2006). Because maintaining existing water quality and instream flows is essential to maintaining and restoring fish populations, a credible plan to maintain existing water quality and instream flows and the data to support it must be part of the environmental review process.

The Plan claims that freshwater habitats downstream of mining activity in streams 2003, 2004 and 2002 will be maintained by providing "minimum base flows" through a "variety of methods and infrastructure" including "sediment ponds, diversion structures, infiltration basins, pumping locations, estimates of water volumes and water-balance information, and runoff control structures." These claims raise a number of concerns given that there has been no comprehensive instream flow study of these streams . First, simply maintaining "minimum base flows" for the decades the mine will be in operation will not maintain downstream fish habitat for salmonids. Baseflow or base runoff is comprised largely of groundwater effluent (USGS 2009). The instream flow needs of salmonids vary by species, stream size, and life stage (Vadas 1999). Spawning salmon require higher flows in small streams than in large rivers. High flows provide the stimulus to coho and Chinook salmon to move upstream to spawning grounds in the Cook Inlet basin (Hayes 2009 and Kerkvliet 2009). Periodic flood flows are also necessary to remove undesirable accumulations of sediment from stream gravels so that it can continue to provide suitable habitat for salmonids and aquatic life (Milhous 1998). Because no long-term instream flow study has been conducted in streams 2002-2004, there is no way to know what flows have to be provided at different times during the year, including high flows during the spawning season, to maintain fish populations, and there is and no way to determine if PacRim's proposed system can provide them.

Second, the complex system proposed to maintain flows presents a substantial risk. One power or equipment failure in the coldest part of the winter could cut off flow to stream 2003, resulting in the loss of fish and other aquatic life. The survival of salmon and other aquatic life in cold climates depends on the continual input of warm groundwater base flow during the coldest winter months. Upstream mining and groundwater pumping will diminish or eliminate this flow. There is also a strong possibility that infiltration basins, settling ponds and bypasses that have been proposed to maintain minimum base flows will freeze up, or deliver during the winter water that is below salmonids' thermal tolerance.

Although infiltration basins have been used in many areas to treat stormwater and recharge groundwater, I was unable to find any studies in the scientific literature documenting the successful use of infiltration basins to maintain groundwater flow to a salmonid steam or to maintain a salmonid steam in a cold climate. The use of infiltration basins to maintain flow to stream 2003 and possibly 2002 and 2004 would be experimental and would be complicated by the fact that the loss of groundwater from these streams would be the result of a cone of depression created by deep mining and pumping to dewater the mine. An independent hydrologist should review the data to make sure that the water from the infiltration basins wouldn't be drawn to the pumps instead of the streams.

Lost Habitat Mitigation:

*Pages 7-8, **Off channel spawning and rearing habitat:*** **The Plan discusses** some of the disadvantages of attempting to create artificial spawning and rearing habitat to compensate for the loss of 17.4 km of high-value spawning and rearing habitat for 5

salmon species, Dolly Varden and Rainbow trout, but there are other disadvantages as well. First, there is a substantial probability that a functional spawning channel couldn't be constructed. It appears that only one successful salmon spawning channel has been constructed in Alaska and an attempt to extend it was not successful (Lachmar et al. 2007). Second, not all species of salmonids found in stream 2003 use artificial spawning channels. Most importantly, unless the equivalent of 17.4 km of new rearing habitat can be created for salmon fry produced in a spawning channel, which does not seem feasible, the fry produced in a spawning channel probably would not survive and would not offset the loss of production from stream 2003.

Pages 8 13. Off Site Mitigation: A number of offsite mitigation measures are proposed, but they do not appear to offset fish losses from mining activities. Any potential benefits of the proposed offsite measures must also be evaluated against the certain loss of 17.4 km of high-value spawning and rearing habitat in stream 2003, and additional potential losses in streams 2002-2004. No such analysis is provided for any of the proposed projects.

Some of the proposed measures -- such as bank stabilization for the Theodore River, constructing a bridge across Threemile Creek, and repair of the Theodore River Bridge, which is owned by an oil company -- offer little or no benefit to fish. Others -- such as restoring Threemile Lake sockeye stocks by eliminating introduced northern pike -- probably have little chance of success because of the limited success of rotenone in removing pike from a system with wetlands, an outlet stream , and ground water inflow.. The proposal to remove the Big Lake dam is puzzling because the dam has already been removed (Ivey 2009).

Replacement of culverts that are blocking upstream fish passage can benefit fish populations if studies show that suitable fish habitat exists upstream of the culvert. There is no indication that any studies have been done to show that replacing these culverts would substantially increase fish habitat. There are already are a number of ongoing programs to identify and fund culvert replacement in the Cook Inlet region. The Department of Fish and Game also has the authority under AS16.05.840 to require the owner of a dam or obstruction to fish passage to remove it.

Page 14: The Plan states that "Technologies and materials now exist that will allow for reconstruction of the 2003 stream channels in a manner that will support pre-mine riverine ecological processes." This statement needs to be supported with examples from the scientific literature of successful salmon spawning and rearing stream/drainage restoration in strip-mined areas. No examples have been provided by PacRim and I have searched the scientific literature and talked to a number of stream restoration experts but have been unable to find any examples.

Similarly, the Plan's statement that Stream 2003 and other Chuitna Coal Mine area streams represent a more comprehensive restoration effort that includes three-dimensional restoration of the entire channel, including floodplain structure and form, is incomplete. Because the entire 2003 aquifer will be altered to a depth of 300 feet or more

there also needs to be a plan to restore the shallow aquifers that supply phreatic groundwater flow to stream 2003 and possibly to streams 2002 and 2004. There is no mention in the Plan of how pre-mining groundwater flow would be restored to these streams. I have also searched the scientific literature and found numerous papers that document how groundwater flow has been altered after strip-mining restoration but no examples of reclamation efforts that have restored groundwater flow to pre-mining conditions

I have similar concerns with the statement that "Relative examples of this type of project can be found in the reclamation of placer-mined streams." First, the damage to salmon streams from alluvial placer mining is very different from strip–mining, which may encompass entire drainages and alter both the surface topography and subsurface geology and the aquifers down to several hundred feet. Restoration projects like the USFS projects on Resurrection Creek have attempted to restore both sinuosity and rearing habitat to a stream impacted by placer mining in the early 1900's by moving and grading spoil piles and providing instream cover. Placer miners channelized the stream and left spoil piles in the flood plain but did not destroy the shallow aquifer that provides groundwater flow to the stream. It appears that the USFS projects have increased rearing habitat for Chinook and coho salmon by connecting formerly isolated channels in mine spoils, but no pre-mining data on fish use is available.

Page 15: The statement that the Chuitna Coal Mine area streams occur in a non-urbanized, pristine watershed where natural functions and processes are intact is true, However, the statement that reclamation design and construction will be largely ***non-structural*** in nature with the exception of log jams and where appropriate and grade controls at intervals in active bed reaches is puzzling. First, strip-mining the entire upper stream 2003 drainage and completely altering the surface and subsurface topography and hydrology will have greater effects on fish habitat than urbanization (May et al 1997). Second, construction of a new stream channel will be structural. ***Structure*** means a complex entity constructed of many parts, and the Rosgen system consists of characterizing and categorizing ***stream structure***. To construct a functioning stream channel using Rosgen's system as the basis, the drainage would have to be recontoured and revegetated (which could take decades or longer), a complex new stream channel constructed, and a new shallow aquifer constructed to provide shallow groundwater flow to the new stream. Problems with the use of Rosgen's system to restore streams are discussed earlier in this report.

Page 16 paragraph 1. It may be possible to design a "stable post mine channel plan form, cross section and profile," but there is no evidence in the scientific literature that a functioning salmon spawning and rearing stream could be reconstructed after the mining proposed for the stream 2003 drainage is completed. The Plan does not provide any examples of where a fully functioning 17.5 km long salmon spawning and rearing drainage and associated watershed -- or anything on a similar scale -- has been successfully restored after strip mining. An extensive literature search by this reviewer hasn't turned up any examples either. Without some evidence of previous success, the

entire concept of using Rosen's classification system to successfully restore a heavily altered drainage is highly questionable.

Page 16. Existing data: The summary of necessary data associated with the stream reclamation is incomplete. To collect information necessary for restoration of the natural hydrological cycles of stream 2003 and the fish populations whose life cycles revolve around seasonal changes in flow and water quality, it is necessary to install stream gauges in strategic stream reaches in stream 2003 and to conduct an instream flow study for a minimum of 5 years or more (Kondolf et al 2000 and Estes and Orsborn 1987). As stated previously, 5 years of flow data is the minimum required by ADNR to grant an instream flow reservation to protect fish habitat to ADF&G (Kline 2009). The number of years of flow measurement would be dependent on knowledge of the fine-scale variation in flows over time and other factors. This has not been done. It is also necessary to measure groundwater flow into stream 2003, because the influx of shallow groundwater is likely the most important factor in overwinter survival of fish and aquatic life (Swales and Levings 1989 and Bennet 2004). Data should be collected over a long enough period of time to determine the full range of daily, seasonal and annual variations in phreatic groundwater flow. This has not been done and this data is not available to design a new functional channel, new shallow aquifers or determine instream flows needs of stream 2002-2004 fish and aquatic life.

Page 20 Beaver activity: Beaver dams affect stream morphology and riparian zones, and provide important habitat for coho salmon and other fishes (Bruner 1989 and Mitchell and Cunjak 2007). The loss of the numerous beaver dams will affect both fish habitat and stream flow in stream 2003. However, this does not appear to be factored into PacRim's stream restoration plans. Beavers will recolonize the upper Stream 2003 drainage only if and when riparian woody deciduous vegetation is fully restored. No estimate is provided of how long reestablishment of woody vegetation may take, but it could be decades or longer. Forty years after peat mining in the Colorado Rockies, sedges and willows that dominated undisturbed sites were largely absent on mined sites (Cooper and MacDonald 2001). Previously reclaimed strip mines in Alaska have been reseeded with grasses and some types of woody vegetation. Monitoring of revegetation at the Usibelli Mine in interior Alaska found that grasses, particularly, boreal red fescue, were the most successful species in part because of the acid soils left after mining (Elliott et al 1986). The floodplain depressions and backwaters PacRim proposes to replace beaver dams and wetlands will not provide the same functions as the 50 beaver dams, 7 lakes, and thousands of acres of wetlands that would be destroyed by mining.

Page 21. Hydrologic and Hydraulic Analysis. Streamflow is the master variable that limits the distribution and abundance of fish species in rivers and streams. Streamflow typically follows a general pattern but may vary hourly, daily, seasonally, yearly and longer. This characteristic pattern of the magnitude, timing, and variability of streamflow defines a river's flow regime. The life cycles of fishes and other aquatic life are inexorably tied to this flow regime. Alteration of any facet of this natural flow regime can result in significant ecological consequences.

Maintaining natural volumes and patterns of instream flow are essential to preserve fish populations and other aquatic life in the unmined lower reaches of stream 2003. Restoration of premining instream flow patterns to a reconstructed stream 2003 drainage would be extremely difficult. I did not find any examples in the scientific literature of restoration of premining in stream flow patterns and volumes to a similarly altered drainage.

Instream flow is used to identify a specific stream flow in cubic feet per second at a specific location for a defined time, and typically following seasonal variations. Instream flows are the stream flows needed to protect and preserve instream resources and values such as fish, wildlife and recreation (Milhous 1998, Vardas 1999, Estes and Orsborn 1987 and Kondolf et al 2000). Instream flow needs are determined by calculations based on several years of streamflow data and from a network of streamflow gauges placed at key locations within a stream, detailed information on depth, velocity and substrate, and a through understanding to the biology of salmonids. (Estes and Orsborn 1987 and Kondolfet al 2000). However, there is no mention in the Plan of the instream flow needs of fish and aquatic life in the streams likely to be affected by the Chuitna Coal Mine, or any indication in the 2006-2008 Chuitna Coal environmental studies reports that an adequate instream flow study particularly in streams 2002 and 2004 and other Chuitna River tributaries which might be effected by hydrological changes from mining has been conducted. There apparently were some stream gauges in the Chuitna drainage during the 1982-84 studies and the Riverside Technologies Hydrology report indicates that there have been up to 4 gauges in stream 2003. However, there is concern that there weren't enough gauges and they were not in the right location to determine instream flow requirements in streams 2002, 2003 and 2004 (Mouw, 2009).

PacRim has confirmed that there has been no USGS stream gauge in stream 2003, and the USGS web site does not show that there are presently any stream gauges in the Chuitna River drainage (USGS 2009). The Fish and Wildlife Protection Plan does not indiciate there are PacRim gauges in the system either, however, the Riverside Technologies Hydrology Component Baseline indicates that there have been four gauges on stream 2003 with 3 to 16 years of data. According to the Fish and Wildlife Protection Plan PacRim proposes to use USGS flood flow regression equations to calculate flood flows for stream design. This may help design channels that will resist flood flows, but it won't provide data necessary to maintain fish habitat.

Page 22 Hydrologic and hydraulic analysis: Although the Plan acknowledges the potential for avulsion and subsequent unzipping of a newly constructed channel due to flooding, it does not acknowledge the length of time that this risk will continue or the potential consequences. Revegetation of the mined areas could take a very long time, and during this time the exposed soils will be subject to erosion. The erosion rate of mined watersheds reported by Haigh (2000) was 1.7 mm-yr.-1, or 85 times greater than erosion at an unmined watershed (.02mm-yr.-1). A 10-, 25- or 100-year flood on an unvegetated hillside and floodplain would not only wash out a newly constructed stream channel, but would carry tons of sediment into the stream channel (Martin and Platts 1981 and

Ruediger and Ruediger 1999). This sediment would wash downstream, smothering spawning and rearing areas in unmined sections of stream 2003 (Cordone and Kelley 1960).

Page 30 paragraph 2: EGL's description of the hyporheic zone as the region below and laterally to the streambed where there is mixing of shallow groundwater and surface water, and its importance in fish spawning and baseflow recharge, is correct However, the role of shallow groundwater and the hyporheic zone in overwinter survival of fish eggs and fry must also be acknowledged. Unfortunately, the Plan does not explain how the flow of shallow groundwater to the hypohreic zone will be restored after excavation of the existing aquifers down to a depth of several hundred feet below the streambed elevation.

Pages 26-33: Typical construction plans and specifications: The techniques described in this section are all bioengineering techniques which have been used to restore damaged streambanks or relatively short sections of relocated streams. After searching the scientific literature and talking to stream restoration experts, I have not been able to find any examples of where any of the techniques proposed in the Fish and Wildlife Protection Plan have been used to restore the ecological functioning of 17.4 km (or even a smaller scale) of a strip-mined salmon spawning and rearing stream, watershed, and associated aquifers. I also reviewed the references used to support the proposed restoration methodology in the Fish and Wildlife Protection Plan and found only two that might provide information on restoration techniques and successful salmon stream restoration projects in strip-mined areas. These were Rosgen, ***Applied River Morphology*** (1996) and Leedy et al ***Environmental Reclamation and the Coal Surface Mining Industry*** (1987).

Rosgen's 1996 steam classification system has been widely used to inventory stream channel topography in watersheds, but it is increasingly drawing criticism from hydrologist and engineers for its use in stream restoration (Simon et al 2007, Gillian 1996, Lacombe and Eaton 2003 and PacRim Coal 2007). Some of the problems with using Rosgen's natural channel design classification system in stream restoration include: (1) alluvial streams are open systems that adjust to altered input of energy and materials, and Rosgen's form-based system largely ignores this component; (2) C5 channels composed of different bank sediments adjust differently and to different equilibrium morphologies in response to identical disturbances, contradicting the fundamental underpinnings of natural channel design and the reference reach approach which PacRim proposes to use to reconstruct a new stream 2003; and (3) Rosgen's system fails to integrate and quantify fluvial processes and channel response. This is important because PacRim has not adequately gauged or conducted instream flow studies on any of the streams likely to be impacted by the project (PacRim Coal 2007).

Although, Rosgen's natural channel design and bioengineering techniques proposed for stream 2003 have apparently not been used to successfully restore salmon spawning streams, they have been regularly used as a cookbook method to restore eroding stream

banks and relocate relatively short sections of streams (Gillian 1996 and Simon et al 2007). One of the difficulties in evaluating the potential effectiveness of stream restoration projects using Rosgen's methodology and bioengineering techniques in restoring streams and fish habitat is that there have been few quantitative scientific post-project studies. One group of scientists who evaluated the success of stream restoration projects using this methodology found that fewer than half had any pre- or post-project monitoring, and of those that did, most monitored only riparian vegetation and not stream habitat (Moeree and Lambert 2004). Another study, which looked at 23,000 restoration projects, found that 34% did not do any post-project evaluation. Of the 70% who said that their projects were successful, 43% either did not have any success criteria or were unaware of what success criteria are (Rumps et al 2007). PacRim did not reference any studies in the scientific literature documenting the use of Rosgen's stream classification system to successfully restore a salmon spawning and rearing stream in a deep strip-mined area, and my searches did not reveal any examples either.

I was unable to locate and review the article ***Environmental Reclamation and the Coal Mining Industry. National Institute for Urban Wildlife, by D Leedy, L. Adams, G. L. Dove, and G. Jones (1985),*** **which is** cited in the Plan. The National Institute for Urban Wildlife, which apparently closed in 1995, sponsored research and wrote papers on urban wildlife problems with limited relevance to the Chuitna Coal Project.

Page 34: Construction Sequencing: Restoring the flow of phreatic groundwater into the hyporheic zone of stream 2003 is essential to restore fish habitat in stream 2003. The only statement in the Plan which may refer to groundwater restoration is that stockpiled soil will be placed in a "manner to replicate premined strata and compacting conditions. This effort will include replication of subsurface confining zones segregating confined and unconfined aquifers."

The Plan does not provide any references to support the methodology they have proposed to use to restore groundwater flow or examples of strip mine reclamation projects where groundwater flow was restored to pre-mining conditions by replacing mining tailings or soil. I was also unable to locate any examples in an extensive search of the scientific literature. However, there is a large body of scientific information documenting long-term disruption of both surface and groundwater flow as the result of recently permitted strip mining and reclamation (Bonta 2007, Wilson 1978 and Schwartz and Crowe 1985). Bonta et al (2007) studied the effect of surface mining and reclamation on physical watershed conditions and groundwater hydrology in three watersheds. This study found that mining disturbances in adjacent watersheds affected groundwater levels in the undisturbed watersheds prior to mining. New subsurface flow paths with different characteristics formed during mining and reclamation. Groundwater recovery in the mined upper saturated zone was slow and irregular both temporally and spatially after reclamation. Wilson (1978) found that the impact of a strip mine can extend far beyond its radius of influence at the water table, and mines near regional discharge areas have a more significant effect on the regional system.

Wetlands and Lakes: There are thousands of acres of wetlands and at least seven lakes and ponds that will all be destroyed by the proposed mine, but there is no discussion in the Plan of the long-term consequences of this impact and apparently no plan to restore these wetlands, lakes and ponds after mining. The importance of wetlands to the productivity of salmon streams is well-documented in the scientific literature (Pess et al 2002 and Pollock et al 2004). A recent study of a stream drainage in Southeast Alaska concluded that "Organic nutrients derived from wetlands comprise the bulk of the stream's water organic nutrient budget on an annual basis." (Hood et al 2008). The loss of wetlands has been correlated with declines in salmon production (Pess et al 2002 and May et al 1997). Even if the physical structure of stream 2003 could be restored, the currently level of productivity probably cannot be because of the permanent loss of the extensive wetlands in this drainage.

Summary:

The plan for stream restoration presented in the Fish and Wildlife Protection Plan is conceptual and few specifics are provided. The goal of the Plan appears to be to construct a new stream channel on top of mine tailings using stream measurements based on Rosgen's 1996 stream classification system. Bioengineering techniques would be used to reconstruct the banks. The floodplain would be revegetated at some later time. Because no comprehensive instream flow study has been conducted on stream 2003, the new channel would be designed based on USGS equations and possibly with comparisons with adjacent drainages. During mining, streamflows to stream 2003 would apparently be maintained through a system of bypasses and infiltration galleries. After reconstruction, PacRim expects stream 2003 to evolve to its premining level of productivity.

There are a number of problems with the Plan. First, there is no scientific evidence that stream 2003 could be restored. PacRim does not provide any examples or scientific documentation to support its claim that stream 2003 can be restored to premining productivity. Scientific studies of mines similar to the proposed Chuitna Coal mine concluded that strip mining for coal may affect groundwater flow over a wide area, including adjacent drainages. Even after restoration as required under federal and state regulations, groundwater flow has been altered from premining conditions. An extensive and lengthy search of the scientific literature and discussions with stream restoration and instream flow experts did not yield one example of where a strip-mined salmon spawning and rearing stream and its associated watershed and aquifer have been successfully restored. A list of the databases searched is included in Appendix 1.

Second, there are problems with the proposed stream restoration methodology. The Rosgen stream classification and associated natural stream channel design methods, which are the basis of the proposed PacRim stream restoration plan, has come under increasing criticism from hydrologists and engineers. Some of the problems with using Rosgen's natural design method in stream restoration include: (1) alluvial streams are open systems that adjust to altered input of energy and materials, and a form-based system largely ignores this component; (2) C5 channels composed of different bank

sediments adjust differently and to different equilibrium morphologies in response to identical disturbances, contradicting the fundamental underpinnings of natural channel design and the reference reach approach which PacRim proposes to use to reconstruct a new stream 2003; and (3) Rosgen's system fails to integrate and quantify fluvial processes and channel response. This is important because PacRim has not adequately gauged or conducted instream flow studies on any of the streams likely to be impacted by the project (Simon et al 2007, Gillian 1996, Lancombe and Eaton 2003 and PacRim Coal 2007).

Third, the on- and offsite mitigation proposed in the Plan would not compensate for the loss of fish habitat in 17.4 km of stream 2003, alteration of groundwater flow which may also affect streams 2002 and 2004, and the diminishment of stream productivity resulting from the permanent loss of thousands of acres of wetlands. If spawning channels could be successfully constructed, they would offset the loss of spawning habitat only for some species in stream 2003, and only if a corresponding amount of rearing habitat could be created to replace the 17.4 km lost in stream 2003, which seems unlikely. The offsite mitigation proposed, such as the proposal to repair a bridge, restore a section of eroding streambank and remove the Big Lake dam (which has already been removed), is of questionable or no benefit.

Fourth, essential studies of instream flow and the input of groundwater to stream 2003 necessary to support salmon and other aquatic life and reconstruct stream 2003 have not been conducted.

Fifth, there are several thousand acres of riparian and upland wetlands in the stream 2002-2004 drainages that probably provide the bulk of stream 2003's annual organic nutrient budget that will be destroyed as the result of mining. However, there is no mention of reconstruction of these wetlands or of the functions they currently provide; n or does the Plan contain any plans to reconstruct them.

H. Conclusions:

There are serious problems with the data collection methodology, analysis and conclusions of the baseline aquatic resources studies and Fish and Wildlife Protection Plan. These include:

1. There are many problems and questions about the data collection methodology, data analysis and conclusions of the 2006-2008 studies. These are identified and discussed in the comments on each report. In addition, it is problematic to base any conclusions on the limited amount of data that has been collected on key issues such as smolt outmigration, adult escapement, winter fish studies, and smolt production between stream systems. Only one year's data has been collected on most of these issues,. Because salmon populations have historically fluctuated greatly over a 20 or more year cycle, the few years of data collected is not sufficient to determine the natural range in salmon populations that would be

affected by the Chuitna coal strip mine. For example adult Chinook escapement into the Deshka River, another upper Cook Inlet spawning stream ranged from 57,939 fish in 2004 to 7,533 in 2008. Juvenile salmon and other forms of aquatic life show similar variability. A minimum of 5-10 years of additional study is necessary to determine the natural range of variability in fish populations that would be affected by the Chuitna Coal Mine. A credible mitigation and restoration plan cannot be developed without adequate data.

2. PacRim has not conducted adequate surface and groundwater studies necessary to: accurately map and quantify the seasonal and long term cycles of groundwater input into streams 2002-2004; determine impacts to the Chuitna River drainage from strip-mining and groundwater pumping associated with mining; provide assurances that groundwater flow to unmined portions of streams 2003 and 2004 can be maintained, or: restore essential phreatic groundwater flow to a reconstructed stream 2003.
3. The uninterrupted flow of shallow groundwater to salmonid spawning streams is essential for overwinter survival of eggs and fry. Strip-mining will interrupt this flow and destroy the shallow aquifers that currently provide groundwater to streams. PacRim has not acknowledged this as an issue, provided a plan to restore groundwater flow, or refereneed any scientific studies showing where an aquifer supplying phreatic groundwater to a salmon spawning and rearing stream has been successfully restored after strip-mining. An extensive search of the scientific literature returned many examples of how strip-mining has altered groundwater flow during and after mining but no examples of where groundwater has been restored to premining conditions.
4. The failure to determine the genetic makeup of salmonid stocks in streams 2002-2003 and the Chuitna River system is a serious deficiency of the Chuitna Coal environmental studies and proposed monitoring program. Data on genetic characteristics of salmon populations are critical for quantifying the status of local reproductive units (demes) and evolutionary significant units. There is mounting evidence that the individual spawning and rearing streams, such as stream 2003, of coho, sockeye and likely other salmonids may be comprised of demes or small locally interbreeding groups that are genetically adapted to the unique conditions in their natal streams. If these streams and the genetically unique salmon demes that use them are destroyed or blocked by strip-mining as proposed, it is unlikely that these local salmon stocks could be restored to their former level of productivity even if a new stream channel could be successfully constructed.
5. It is probably not possible to reconstruct a new stream with the same level of productivity as the current stream 2003. PacRim has not provided any examples of where a strip-mined salmon spawning and rearing drainage the size of stream 2003 (17.4 km) has been restored to premining productivity. An extensive search of the scientific literature and discussions with stream restoration experts in Alaska and elsewhere has also not produced any examples.
6. There are problems with PacRim's plan to use Rosgen's 1996 Applied River Morphology as the basis for stream 2003 reconstruction. Rosgen's 1996 stream classification system has been widely used to inventory stream channel

topography in watersheds, but it is increasingly drawing criticism from hydrologists and engineers for its use in stream restoration. Some of the problems with using Rosgen's natural channel design classification system in stream restoration include: alluvial streams are open systems that adjust to altered input of energy and materials and Rosgen's form-based system largely ignores this component; Rosgen C5 channels composed of different bank sediments adjust differently and to different equilibrium morphologies in response to identical disturbances, contradicting the fundamental underpinnings of natural channel design and the reference reach approach that PacRim proposes to use to reconstruct a new stream 2003,; and Rosgen's system fails to integrate and quantify fluvial processes and channel response. This is important because PacRim has not adequately gauged or conducted instream flow studies on any of the streams likely to be impacted by the project.

7. Even if Stream 2003 could be successfully restored to full physical and ecological function, it may not be possible to restore it to its former level of biological productivity because of the loss of marine derived nutrients (MDN) from salmon carcasses and the permanent removal of all the wetlands in the mine area. Wetlands and MDN are the primary sources of stream nutrients and productivity in salmon streams.
8. The offsite mitigation offered in the plan (i.e. removal of the Big Lake Dam, bridge repair etc.) has little or no potential to offset the loss of fish populations and 17.4 km of high-value fisheries habitat that would be destroyed or altered by mining.
9. There is a good chance that the spawning channels offered as onsite mitigation for the loss would not be successful. Even if these channels were used by spawning adults, any fry produced in spawning channels would probably not survive unless all of the high-value rearing habitat that would be destroyed or blocked by could also be created off site which is very unlikley.

The information provided in the PacRim Coal 2006-2008 Environmental Studies and the summary of the 1980's studies is inadequate to determine the effect of the proposed Chuitna Coal Mine on fish and fish habitat in the Chuitna drainage. It is also inadequate to develop a mitigation plan and a restoration plan for fish populations and habitat that would be impacted by mining. Additional research is needed. Appendix 2 contains a list of research needs identified during this review.

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Appendix 1: Databases Searched

Academic Search Premier
Alaska and Polar Periodicals
Aquatic Sciences and Fisheries Abstracts
Conference Papers Index
Environmental Sciences and Pollution Management
Fish and Fisheries Worldwide
Science Direct
Water Resources Abstracts
Wildlife and Ecology worldwide
Google Science
Transactions of the American Fisheries Society

Appendix 2: Additional Studies Needed

- 5-10 years of sonar and/or weir counts of adult and salmon smolt in the mainstem Chuitna River and streams 2002-2004.
- Genetic analysis of salmonid populations in the mainstem Chuitna River and streams 2002-2004.
- Determine if the Chuitna River and streams 2002-2004 are rearing or spawning limited for all species of salmonids.
- Use remote sensing to identify and map groundwater input into streams 2002-2004 and the Chuitna River.
- Determine the relationship between, and the importance of, groundwater and surface water sources in streams 2002, 2003 and 2004 and relate this to the use of available habitat for incubation, rearing, and overwintering.
- A comprehensive study of marine derived nutrients and terrestrial/wetlands nutrient cycling in the Chuitna River drainage and streams 2002-2004.
- A comprehensive study of freshwater and marine survival rates of Chuitna River salmon populations using coded wire tags and other techniques.
- A comprehensive multiyear study of fish movements within and between Chuitna River tributaries using coded wire tags.
- A comprehensive study of factors, including groundwater, affecting overwinter survival of fishes in Streams 2002-2004. Characterize and quantify overwintering habitat.
- Study juvenile salmon rearing and smolt outmigration in the lower Chuitna River and nearshore waters of Cook Inlet.
- Study the effect of weirs on upstream fish movements in streams 2002-2004.
- A comprehensive study of the overwintering behavior and habitat factors (e.g. groundwater, water temperature, instream flow etc.) affecting the overwinter survival of salmonids, other fishes and aquatic life in streams 2002-2004.
- Locate fish spawning and overwintering areas within streams 2002-2004 and the Chuitna River. Measure hyporheic and phreatic groundwater flow and water temperatures in these areas.
- Study the use of the mainstem Chuitna River as rearing habitat by coho and Chinook salmon.
- Conduct a two-year field study to determine the feasibility of using infiltration galleries to restore streamflow to a stream, especially during the winter months.
- Study the rates at which Chuitna salmonids return to natal and non-natal areas to spawn and rear.
- Determine egg to emergent fry and emergent fry to smolt survival rates in streams 2002-2004 and the Chuitna River. Identify and quantify freshwater mortality factors (e.g. predation, freeze-out, etc.).
- Determine the age and size at which Chuitna River coho and Chinook salmon smolt, using coded wire smolt tags and adults recoveries.
- Determine how and when high water events initiate upstream Chinook and coho spawning movements in the Chuitna River drainage.
- Quantify how floods and high water events affect the accuracy of weir counts of upstream and downstream movements of fish in streams 2002-2004.

EXHIBIT 18

Report on Chuitna Coal Project Aquatic Studies and Fish and Wildlife Protection Plan

Prepared by Lance Trasky, Lance Trasky and Associates
June 1, 2009

Summary of Conclusions:

There are serious problems with the data collection methodology, analysis and conclusions of the baseline aquatic resources studies and Fish and Wildlife Protection Plan. These include:

1. There are many problems and questions about the data collection methodology, data analysis and conclusions of the 2006-2008 studies. These are identified and discussed in the comments on each report. In addition, it is problematic to base any conclusions on the limited amount of data that has been collected on key issues such as smolt outmigration, adult escapement, winter fish studies, and smolt production between stream systems. On most of these issues, only one year's data has been collected. Adult salmon returns may vary by as much as 700% over a ten year period in Cook Inlet. Juvenile salmon and other forms of aquatic life show similar variability. A minimum of 5-10 years of study's study is probably necessary to determine the natural range of variability in fish populations that would be affected by the Chuitna Coal Mine. A credible mitigation and restoration plan cannot be developed without adequate data.
2. PacRim has not conducted adequate surface and groundwater studies necessary to: accurately map the input of groundwater into streams 2002-2004; determine impacts to the Chuitna River drainage from strip-mining and groundwater pumping associated with mining; provide assurances that groundwater flow to unmined portions of streams 2003 and 2004 can be maintained; or restore essential phreatic groundwater flow to a reconstructed stream 2003.
3. The uninterrupted flow of shallow groundwater to salmonid spawning streams is essential for overwinter survival of eggs and fry. Strip-mining will interrupt this flow and destroy the shallow aquifers that currently provide groundwater to streams. PacRim has not acknowledged this as an issue, provided a plan to restore groundwater flow, or referenced any scientific studies showing where an aquifer supplying phreatic groundwater to a salmon spawning and rearing stream has been successfully restored after strip-mining. An extensive search of the scientific literature returned many examples of how strip-mining has altered groundwater flow during and after mining but no examples of where groundwater has been restored to premining conditions.
4. The failure to determine the genetic makeup of salmonid stocks in streams 2002-2003 and the Chuitna River system is a serious deficiency of the Chuitna Coal environmental studies and proposed monitoring program. Data on genetic characteristics of salmon populations are critical for quantifying the status of local reproductive units (demes) and evolutionary significant units. There is mounting evidence that the individual spawning and rearing streams, such as stream 2003,

of coho, sockeye and likely other salmonids may be comprised of demes or small locally interbreeding groups that are genetically adapted to the unique conditions in their natal streams. If these streams and the genetically unique salmon demes that use them are destroyed by strip-mining as proposed, it is unlikely that these local salmon stocks could be restored to their former level of productivity even if a new stream channel could be successfully constructed.

5. It may not be possible to reconstruct a new stream with the same level of productivity as the current stream 2003. PacRim has not provided any examples of where a strip-mined salmon spawning and rearing drainage the size of stream 2003 (17.4 km) has been restored to premining productivity. An extensive search of the scientific literature and contacts with stream restoration experts in Alaska and elsewhere have not produced any examples either.
6. There are problems with PacRim's plan to use Rosgen's 1996 Applied River Morphology as the basis for stream 2003 reconstruction. Rosgen's 1996 stream classification system has been widely used to inventory stream channel topography in watersheds, but it is increasingly drawing criticism from hydrologists and engineers for its use in stream restoration. Some of the problems with using Rosgen's natural channel design classification system in stream restoration include: alluvial streams are open systems that adjust to altered input of energy and materials and Rosgen's form-based system largely ignores this component; Rosgen C5 channels composed of different bank sediments adjust differently and to different equilibrium morphologies in response to identical disturbances, contradicting the fundamental underpinnings of natural channel design and the reference reach approach that PacRim proposes to use to reconstruct a new stream 2003,; and Rosgen's system fails to integrate and quantify fluvial processes and channel response. This is important because PacRim has not adequately gauged or conducted instream flow studies on any of the streams likely to be impacted by the project.
7. Even if Stream 2003 could be successfully restored to full physical and ecological function, it may not be possible to restore it to its former level of biological productivity because of the loss of marine derived nutrients (MDN) from salmon carcasses and the permanent removal of all the wetlands in the mine area. Wetlands and MDN are the primary sources of stream nutrients and productivity in salmon streams.
8. The offsite mitigation offered in the plan (i.e. removal of the Big Lake Dam, bridge repair etc.) has little or no potential to offset the loss of fish populations and 17.4 km of high-value fisheries habitat that would be destroyed by mining.
9. There is a good chance that the spawning channels offered as onsite mitigation for the loss would not be successful. Even if it were, however, any fry produced in spawning channels would probably not survive unless the equivalent of 17.4 km of high-value rearing habitat could also be created.

The information provided in the PacRim Coal 2006-2008 Environmental Studies and the summary of the 1980's studies is inadequate to determine the effect of the proposed Chuitna Coal Mine on fish and fish habitat in the Chuitna drainage. It is also inadequate to develop a mitigation plan and a restoration plan for fish populations and habitat that would be impacted by mining. Additional research is needed. Appendix 2 contains a list of research needs identified during this review.

EXHIBIT 19

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I. Introduction

This is a report from the review of the following documents: 1) "Aquatic Biology: Existing Information for the Chuitna Coal Project", June 18, 2006, prepared by OASIS Environmental Inc., 2) "Chuitna Coal Project 2006 Freshwater Aquatic Biology Study Program" April 4, 2007, prepared by OASIS Environmental Inc., 3) "Chuitna Coal Project – 2007 Freshwater Aquatic Biology Study Program", March 2007, prepared by OASIS Environmental Inc., 4) Chuitna Coal Project Wetland Functional Assessment", March 5, 2008, prepared by HDR Alaska Inc., 5) "Part D7 Fish and Wildlife Protection Plan, Chuitna Coal Project", July 2007, prepared by PacRim Coal, LP, 6) "Chuitna Coal Project Hydrology Component Baseline Report, Historical Data Summary", March 2007, prepared by Riverside Technology Inc., 7) "Winter Freshwater Fish Habitat Baseline Report, Chuitna Coal Project", January 2009, prepared by OASIS Environmental Inc., and 8) "Movement and Abundance of Freshwater Fish in the Chuit River Drainage, Alaska, May through September 2008", February 18, 2009, prepared by LGL Alaska Research Associates, Inc.

This scientific review found critical elements missing from the documents. The failure to address food webs, trophic linkages, interactions among upstream-downstream, stream-riparian, stream-marine, and basin-wide linkages (below and above ground) severely undermines the ability of the mining plans to protect ecosystem function during mining and to restore it post-mining. Further, the lack of consistent, long-term sampling (at least 10 continuous years via consistently applied sampling techniques) does not provide the needed estimate of annual biological variability or range of variability, and therefore does not provide a reference to which post-mining rehabilitation effects can be compared. Finally, recreating the structural complexity and interconnectivity of the below-ground sediment layers in the back-filled mine pit will be impossible, permanently and negatively affecting the natural flowpaths and hyporheic function (including natural upwelling and downwelling) upon which existing biological productivity and biocomplexity depend.

II. General comments

A. Food webs

Noticeably missing from the existing baseline studies and restoration plans in the Chuitna system is an understanding of food webs, trophic processes, and their dynamics. The flow of nutrients, detritus, and prey to and through food webs has been well studied in recent years, and it is clear that food web complexity and productivity is what in part drives fish and other aquatic consumer populations (Polis et al. 2004, Wipfli and Baxter In review).

In baseline studies in the Chuitna system, benthic macroinvertebrates (BMI) and fish were sampled for a few years, independent of each other (i.e., there was no attempt to understand their trophic interactions and their dependence upon and interactions with their environment), but the role that BMI play in supporting fish populations, when and where they are important, and to which fish species and ages classes has been completely overlooked. And BMI are just one small part of the larger food base that supports fish. The science on food webs has progressed markedly over the past several years (Polis et al. 2004), including that from Alaska (Wipfli In press). BMI, along with many other sources of nutrients and prey, are critical to the survival and production of fish, including juvenile salmonids (Wipfli and Baxter In review). Understanding the trophic pathways in the Chuitna system is one of the first steps necessary for restoration planning and monitoring, and these studies and analyses need to be undertaken. Further, sampling of trophic pathways can and should be done for a minimum of 10 years to begin to get an understanding of the spatial, and seasonal and annual variability in prey abundance and prey flow in these food webs.

B. Watershed-scale perspective

Understanding *watershed-wide* processes, linkages, and interactions is also key to understanding what drives ecosystem productivity and ultimately fish populations, and should therefore be a central part of understanding ecosystem function and its restoration in the Chuitna system. Multiple spatially and ecologically distinct but interacting habitats throughout the drainage ultimately drive fish demographics and productivity, within the more limited and confined regions of fish inhabitation in the watershed. In other words, what happens in one part of the drainage can influence other parts of the drainage, e.g., headwater effects on downstream habitats and communities (Vannote et al. 1980, Gomi et al. 2002, Wipfli and Gregovich 2002, Wipfli et al. 2007, Wipfli and Baxter In review). This has been particularly highlighted for some of Alaska's watersheds. Understanding large wood budgets, and nutrient, organic matter, and prey budgets at the watershed scale would help determine the feasibility of restoration plans.

There has been very little, if any, thought or sampling devoted to understanding broader watershed-scale processes and their interactions across time and space in the Chuitna system (e.g., how headwaters affect downstream habitats, or how riparian

zones affect the flow of terrestrial invertebrates to fishes). This clearly needs to be undertaken if there is any hope of protecting the Chuitna system during mining operations and restoring the Chuitna system after mining has taken place. As indicated in the above section II. A., this should be accomplished over the course of at least 10 years to get a minimal measure of annual variability.

C. Off-channel-stream linkages

Off-channel habitats such as wetlands, ponds, remnant oxbow channels, etc., can be important rearing, overwintering, and foraging areas for fish (Limm and Marchetti 2009). Understanding the role/function of these off-channel habitats is crucial to understanding broader watershed function. These habitats were not adequately sampled for fish or invertebrates during the baseline sampling, nor was any attention given to foraging ecology of fish and determining what prey are responsible for supporting the various species of fish, and when (e.g., diet analysis), in these habitats. Maintaining the productivity of off-channel systems, and their hydrological connectivity to the larger channels is crucial for the restoration of long-term productivity of the Chuitna system.

Beaver dams are an important component of broader riverine function and productivity. For example, the habitats they create provide key rearing, foraging, and overwintering habitat for fish. The function of beaver pond habitats, and their role in broader watershed productivity, needs to be better understood in the Chuitna system, and more care needs to be given to protecting these habitats, and allowing them to persist at current levels during and after mining operations.

D. Riparian-aquatic linkages

The science on the interactions between streams and their riparian zones has progressed substantially over the years, particularly with our understanding of the flow of food (prey - largely invertebrates) and nutrients between these two habitat types (Richardson et al. In press). Ecologists have long recognized the importance of terrestrial inputs of nutrients, dissolved organic carbon, and plant matter to microbial productivity and invertebrate production in streams (Cummins 1974; Vannote et al. 1980). While these nutrients and energy indirectly feed higher consumers such as salmonid fishes, terrestrial invertebrates that fall into streams are a relatively high quality food source that is directly available to fish. Terrestrial invertebrate subsidies to streams can be substantial during the plant growing season, with annual inputs to forested temperate streams as high as 11×10^3 mg m^{-2} yr^{-1} (see Baxter et al. 2005 for a review), dependent in part on the extent and composition of riparian vegetation (Mason and MacDonald 1982; Edwards and Huryn 1995; Romero et al. 2005). Terrestrial invertebrates can comprise more than half of the energy ingested by stream fishes (Wipfli 1997; Allan et al. 2003) and are often the preferred prey of juvenile salmonids (see reviews by Hunt 1975, Baxter et al. 2005). Wipfli (1997) found that terrestrial prey inputs averaged 10 mg dry mass m^{-2} d^{-1} but at times were as high as 39 mg dry mass m^{-2} d^{-1}, and comprised half or more of the energy intake by salmonids in

small streams in southeastern Alaska. In other places, terrestrial invertebrates have been shown to comprise roughly half or more of the annual prey ingested by salmonids (Kawaguchi and Nakano 2001; Nakano and Murakami 2001), with significant consequences for fish growth and abundance (Baxter et al. 2007; Kawaguchi et al. 2003).

Vegetation type (e.g., plant species, coniferous vs. deciduous) and plant community structure affects the amount and type of food supporting salmonid populations (Piccolo and Wipfli 2002, Allan et al. 2003, Richardson et al. In press). Deciduous species such as alder supply more food than many conifers in Alaska (Piccolo and Wipfli 2002, Allan et al. 2003).

Understanding the process of energy flow from streams to riparian habitats, feeding riparian consumers such as birds, and of prey flow from riparian habitats to streams is crucial in the Chuitna system. Knowing how the current plant community affects trophic pathways and food supplies that flow from riparian habitats to streams is essential to more fully understand how this particular trophic process is driving fish populations in the Chuitna system currently. It is also essential to understanding how these processes need to be protected and restored in post-mining reclamation activities, to ultimately restore and sustain fish productivity and other consumers.

E. Headwater-downstream linkages

Ecological linkages between headwater streams and larger-order rivers are better understood today than they were years ago, and a growing body of literature indicates that headwaters are crucial for sustaining the structure, function, productivity and biocomplexity of the downstream ecosystems they flow into (Haigh et al. 1998). Headwater streams provide downstream habitats with a multitude of ecosystem services, including water, nutrients (e.g., nitrogen and phosphorus), food (e.g., organic matter and invertebrate prey for fishes, salamanders, insectivorous birds), and wood which provides structural habitat for biota (Wipfli and Gregovich 2002, Compton et al. 2003, Gregory et al. 2003, Wipfli et al. 2007). They also serve as refugia, spawning habitats, and source areas for biodiversity (Bramblett et al. 2002, Meyer et al. 2007).

Headwater streams affect the ecology and biological integrity of downstream reaches for several reasons. First, headwater habitats encompass the bulk of stream networks and watershed land areas (Naiman 1983a, Benda et al. 2005). Second, they are closely tied spatially to larger streams, entering these waters at numerous contact points (tributary junctions) along channel networks (Leopold et al. 1964). And third, the water draining from headwater streams provides a continual source of essential products (e.g., nutrients, food, wood), which support aquatic and riparian biota (Naiman & Sedell 1979, Naiman 1982, 1983b, Meyer and Wallace 2001, Wipfli and Gregovich 2002). Further, in coastal Alaska and the Pacific Northwest, higher gradient landscapes can often provide fast flowing headwater channels, speeding the delivery process of materials from steep gradient headwater areas to the receiving

food webs downstream. Headwater tributary streams, including those that are fishless, are sources of invertebrate prey to predatory fishes in recipient mainstem habitats.

Studies have shown that organic matter and nutrients originating upstream can fuel secondary production in downstream habitats (Vannote et al. 1980; Wallace et al. 1997), but more recent investigations have focused on the transport of invertebrate prey down stream networks, subsidizing downstream fish-bearing food webs via fluvial transport of invertebrates (Wipfli et al. 2007). Wipfli and Gregovich (2002) showed that fishless headwaters may be a year-round source of invertebrates to fish habitats lower in drainages of SE Alaska. They calculated that within a typical southeastern Alaska watershed, subsidies from fishless headwaters (163 mg dry mass of invertebrates $stream^{-1}$ d^{-1}; 10.4 g dry mass of detritus $stream^{-1}$ d^{-1}) are at levels that could theoretically support 0.2 to 2.0 young-of-the-year coho fry per m^{-2} of stream reach downstream. In addition, Piccolo and Wipfli (2002) reported that headwater subsidies of both invertebrates and leaf litter were strongly mediated by upland forest management and riparian vegetation type. Further, prey from headwater sources appear to be seasonally important at fishless-fish habitat interfaces, as some fish species may seek out these specific habitats during certain times of the year (Bramblett et al. 2002; Bryant et al. 2004), and may partly explain the high fish densities often seen at tributary junctions (Benda et al. 2004).

Recognizing the importance of headwater-downstream linkages and at least attempting to understand the strength of these linkages in the Chuitna system is critical for protecting its short and long-term ecosystem productivity, biodiversity, and complexity throughout the entire watershed.

F. Marine-freshwater linkages

The science on ecosystem linkages between the ocean and fresh water has progressed remarkably over the last decade, and much of this new science has been generated from Alaska. Strong ecological linkages connect marine systems and watersheds via runs of anadromous fishes (e.g., salmon, stickleback), and these linkages have been shown to be essential to the long-term productivity and sustainability of riverine function, nutrient supply and storage, and food web and fish productivity (Wipfli et al. 1998, Gende et al. 2002, Naiman et al. 2002, Wipfli et al. 2004). These linkages still exist naturally and are strong in Alaska because Alaska still supports good runs of salmon, but riverine habitat degradation and loss can severely impact these linkages. In the Pacific Northwest, suppressed salmon runs have led to severe nutrient deficits, leading to reduced freshwater productivity and serious problems with attempts to restore salmon and their ecosystems (Gresh et al. 2000).

Marine inputs from adult salmon returning to freshwater habitats to spawn provide this linkage through providing major energy and nutrient subsidies to freshwater food webs (Krohkin 1967, Krohkin 1975, Mathisen et al. 1988, Elliott 1997; Schmidt et al. 1998; Wipfli et al. 1998; Scheuerell et al. 2005). Salmon range from 2-50 kg each

and often return in very high numbers. Returning adult salmon densities can range broadly from a few spawners per stream, to many spawners m^{-2} (> 20 kg m^{-2}) in cases where upstream fish passage is partially or fully blocked. Even very small watersheds can experience large returns of salmon for their size. Following salmon runs, marine carbon, nitrogen, and other nutrients are sequestered at multiple trophic levels in freshwater and terrestrial ecosystems (Kline et al. 1990; Chaloner et al. 2002a; Hicks et al. 2005), and these subsidies dramatically elevate aquatic productivity. The amount of basal trophic levels such as biofilm and invertebrates can increase up to 25 times in Alaska stream systems that receive spawning salmon, and more salmon lead to increased responses, up to a point (Wipfli et al. 1999, 2003). Increases in benthos can be so high simply because Alaska's watersheds, particularly those along the coast, tend to be nutrient limited (mainly by orthophosphate), so any amount of nutrients added, in this case via returning salmon, help sustain ecosystem productivity and function.

Further, aquatic invertebrates colonize and consume salmon carcasses (Piorkowski 1995) and can be found in high densities on and around dead salmon soon after the run (Minakawa and Gara 1999; Chaloner and Wipfli 2002; Claeson et al. 2006). Chaloner et al. (2002b) showed invertebrate growth rates increase in the presence of carcasses in Alaska streams. Through invertebrate prey production these increases are translated to stream-resident salmonids, and young fish also ingest salmon tissue and eggs directly (Bilby et al. 1996). Wipfli et al. (2003) showed that stream-resident salmonids grew faster and larger in southeastern Alaska streams enriched with salmon carcasses and eggs. Similarly, in selected Washington streams, juvenile salmonid growth increased in response to artificially-added salmon carcasses (Bilby et al. 1996; 1998).

Further, salmon runs are important for supplying food webs with omega-3 fatty acids, lipids, proteins, and other macromolecules that are essential for the integrity and health of stream food webs and fish (Heintz et al. 2004, Wipfli et al. 2004, Heintz et al. In review). Juvenile coho salmon rearing in streams with more salmon sequester more energy, put that energy into body reserves that in turn increase their likelihood to survive and eventually return from the ocean to spawn.

No baseline work was done to understand the extent of these ocean-watershed linkages and the marine subsidy effects on the Chuitna system and its food webs. This work needs to be completed for the Chuitna. Like other Alaska systems, the Chuitna likely depends greatly upon the annual returns of salmon, and slumps in returns (from mining impacts for example) could easily lead to long-term and potentially permanent losses of salmon in the system, as a result of long-term nutrient deficits as seen in parts of the Pacific Northwest (Gresh et al. 2000). Simple and somewhat routine measurements of stable isotopes, fatty acid signals, and lipid content of selected BMI and fish, for starters, would tell us a lot about how and to what extent the Chuitna system depends upon the marine system to sustain its health and productivity. This baseline information is important for rehabilitation, as it

establishes the degree to which the Chuitna system relies upon marine nutrients (e.g., nitrogen) and energy (i.e., carbon).

G. Physical-chemical-biological interactions

1. Hydrologic flowpaths and linkages. Hydrologic flowpaths that currently exist, which are crucial to biological activity, food web productivity, hyporheic processes and exchange of materials with streams (Stanford and Ward 1993, Stanford et al. 2005, Winter 2007), will be destroyed in the Chuitna system from the coal mining as proposed, and cannot be recreated. And riverine systems also cannot 'repair' such damage. The mosaic of sorted sediment veins and networks (Cardenas et al. 2004, Packman et al. 2006) that have been laid down over thousands of years through natural channel evolution and development along the riverine corridor (including deep below-ground habitats) makes the Chuitna system the fish producer that it is today. Removal of the stream channel and its associated hydrologic system during mining operations will permanently disrupt ground water movement associated with the stream, hyporheic exchange, water chemistry, and stream water temperatures for the life of the stream. This will greatly and irreparably impact the Chuitna food web, its productivity, and could single-handedly destroy the Chuitna system as a viable salmon producer. It will also likely destroy the Chuitna system for other aquatic and riparian species that rely on its intact stream, pond, lake, and wetland habitats.

Also, it is essential that even the smallest tributaries remain hydrologically connected (surface and subsurface) to the larger channels during and after the mining activities. These tributaries are travel corridors and seasonal refugia for aquatic species, including invertebrates and fish, and are key components contributing to the overall health, function, and productivity of the Chuitna system.

2. Contaminants. If increases in flocculants and metal concentrations, and changes in pH occurs, which is often the case during and after mining, loss of in-stream invertebrate production will occur depleting prey resources for fishes (Rosenberg and Resh 1993, Loeb and Spacie 1994). This may be a problem during and long after the mining is completed, and particularly important for the mining void after it is backfilled, as water fills and drains the sediment-filled pit. As in the above point, water chemistry degradation could single-handedly devastate food web integrity, complexity, and productivity, impacting fish for the life of the Chuitna system. Rearing salmonids in freshwater systems in Alaska are often food limited, and any reduction of the natural flow of prey (abundance, species richness, seasonal timing, nutritional value) is expected to affect fish abundance and productivity.

Flocculants can both contaminate stream water, and physically impair aquatic habitats, all but eliminating suitable substrate and a living environment for BMI. The flocculent often covers streambeds, stream gravels and woody debris, eliminating essential habitats and making them inaccessible substrate for BMI. Loss of water quality, even at small increments, has been well documented to devastate BMI species, particularly mayflies, stoneflies, and caddisflies (Loeb and Spacie 1994).

These groups of invertebrates have been shown to be important in the Chuitna system as pointed out in the Chuitna baseline monitoring studies (OASIS 2007, OASIS 2008), and loss of these species may mean a major loss of food resources for fish. Loss of food will mean loss of fish.

3. Annual variability. Noticeably missing from the baseline data collection from the Chuitna system – as alluded to earlier in this document – are estimates of long-term annual variability, including for water chemistry and quality, water temperature, physical characteristics of the stream, streambed and riparian zone, and biological parameters of the drainage. An adequate baseline sampling program should encompass a long enough period of time to: 1) capture the range of natural variability, 2) provide a reliable estimate of the true population mean, and 3) provide a benchmark (target) for rehabilitation efforts. As an example, Adkison and Finney (2003) show how salmon abundance can vary widely over long time periods (e.g., Figs 2 and 4 in their paper -- Salmon "catch" is often used as an index of salmon abundance). Salmon can easily range 10-20x in population size, and their high-low population cycles appear to recur, in this case every 5-10 years, embedded within much longer time scales of nearly 50 yrs. Although the Adkison and Finney (2003) examples represent salmon, they are a good model for other species, and may further provide some expectations about variability in physical and chemical components of natural systems as well. Limited baseline data of only a few years in the Chuitna system does not reflect the range of natural variability, or what the true means are, regarding stream flow, chemistry, and invertebrate and fish densities. In other words, we don't know where, on figures 2 and 4 (Adkison and Finney 2003), the current biological population densities are in relation to the baseline data that has been collected from the Chuitna system (i.e., with invertebrates, fish, etc). Present population densities could be near the low or high range of natural variability, or somewhere in between. Because of the limited baseline data period, it is impossible to say what the true rehabilitation targets and restoration expectations should be for Pac Rim.

The few years that the biological data were collected tells us little about the natural range of variation, which needs to be understood in order to put post-mining restoration effects and monitoring into context. A minimum of 10 contiguous years of chemical, physical, and biological sampling is needed to begin to get an understanding of the bigger picture of annual variability, and how natural variability drives Chuitna biological populations and communities.

H. Fish

1. Homing of adult salmon. Olfactory cues with returning salmon and changes in water chemistry from mining operations during and after mining could be a major issue (Royce and Watson 1987, Groot and Margolis 1991), and has not been at all addressed. Even slight changes in pH or metal concentrations could change the homing behavior and therefore returns of adult salmon to the Chuitna system.

2. Displacement mortality. Moving fish from the mining area to other habitats will be a problem for fish, as their densities will undoubtedly exceed local habitat carrying capacity, causing high mortality in moved fish and even in existing locally-resident fish. Research in Alaska has shown that stream-dwelling salmonids are food limited, so any increase in local fish densities will most certainly result in fish stress and mortality (Wipfli 1997, Wipfli et al. 2003). Competition for food and space is expected to lead to indirect mortality, as well as direct mortality from predators as displaced fish search for new, desirable habitats. There is no evidence in the literature suggesting otherwise. Further, if food is limiting, then the upstream-downstream impacts highlighted in section II. E. are expected to apply here, where impacts in the mined area will affect habitats downstream (Vannote et al. 1980, Gomi et al. 2002, Wipfli and Gregovich 2002, Wipfli et al. 2007) by affected material that gets transported downstream (e.g., food).

3. Demographics and ecology over broader temporal and spatial scales. Also needed are estimates of fish abundance (especially juvenile and spawning adult salmon), distribution, seasonal use of habitats, and fish movement throughout the Chuitna system over a longer time period. A watershed is a highly interconnected complex of aquatic (and terrestrial) habitats, including mainstems, off-channels, beaver ponds, wetlands, and lakes. This collection of habitats serves as refugia, foraging areas, movement corridors, and overwintering habitats throughout different times of the year. Use of all these habitats over broader spatial (watershed) and temporal (10-year) scales will provide a needed, more complete picture of habitat use and variation through time. This will provide a more realistic chance of detecting mining impacts. A few years of inconsistently collected data, as is currently the case, is insufficient. More years of pre-impact data will also improve the robustness of the BACI design proposed by LGL (LGL 2009). While the BACI design may be appropriate if there were sufficient data, four years of pre-impact data is inadequate. And, to improve the BACI approach, both 2002 and 2004 should be used as reference streams (Smith 2002). With that said however, a full BACI assessment should be performed by a statistician familiar with BACI designs.

4. Salmon genetics. A salmon geneticist should be consulted regarding the potential loss of a genetically distinct salmon subpopulation. If the spawning population of salmon present in the Chuitna system is genetically distinct from neighboring populations, this subpopulation may be permanently lost or severely depleted, preventing recolonization of salmon during and after the mining activities.

I. Invasive species

Invasive species continue to spread and impact aquatic ecosystems throughout the globe (Mooney and Hobbs 2000). Invasive species often invade when human traffic enters or increases in a particular place, increasing the likelihood that an animal life stage or plant reproductive part is brought into a given place. Invasives often spread rapidly and displace and impact native species, in both terrestrial and aquatic systems (Mooney and Hobbs 2000). Alaska has been spared many of the problems associated

with invasive species because these plants and animals have simply not been introduced in numbers as in the lower 48 states and other places, but there are several invading species that have taken hold in Alaska, especially near population centers such as Anchorage.

There is a substantial risk that one or more invasive species could get introduced and spread in the Chuitna system. Pike are of particular concern, as they can devastate young salmon populations. Precautions need to be taken to prevent introductions and spread, and a clearly articulated plan needs to be in place that describes how invasive species introductions will be prevented, how monitoring for them will be undertaken, and what will be done to eradicate an invasive species if it does get introduced and established.

J. Other

1. Case examples. PacRim must be required to provide examples of a stream that has experienced a mining operation of this nature and scale, which has been successfully restored to its pre-mining ecosystem function, including trophic processes. The local and small-scale rehabilitations that have taken place in other places have had varied levels of success (Palmer review), but nothing has been on the scale of this proposed project. At this point, there is no evidence this rehabilitation effort can be successful, particularly given the scale of this project and the massive and wide-scale disruption of the natural flowpaths in the Chuitna system from the proposed mining activities.

2. Floods. The documents reviewed do not address the risk of a blowout from a large flood event. The entire stream is subject to severe erosion if one of the rehabilitation sites is lost during a large unanticipated flood event, causing that and downstream habitats to unravel. This will not only devastate the physical nature of the downstream portions of stream 2003, but will greatly reduce food web productivity and complexity, leading to high mortality in fish and other riparian and aquatic consumers that rely on aquatic and riparian invertebrates, and will almost certainly impact the river down to salt water. A detailed plan must be in place to deal with such an event.

3. Mine pit void. How will the void left behind from removing 140 vertical feet of coal be filled? How will layers be compacted adequately (not too much, not too little) when the overburden is put back in the mined pit as to not restrict flow and flowpaths, and to avoid slumping? Slumping seems inevitable as replaced sediments settle and conform through time, and this will completely and permanently change the naturally complex hydraulic nature of the Chuitna system, in turn affecting hyporheic exchange, water chemistry and temperature, and therefore riverine food webs, well beyond our lifetimes.

4. Sensitive, rare and endangered species. To ensure accurate identification of all sensitive, rare and endangered species, the existing collections made by PacRim (i.e., invertebrates) should be carefully inspected by taxonomic specialists, and wetland

sampling should be greatly expanded (spatially and temporally) for all plant and animal species.

5. Revegetation. How will the fabric used during stream bank rehabilitation allow for natural seeding and germination? How can/will seeds penetrate the fabric?

6. Incomplete stream restoration plan. The *Fish Protection Plan* is terribly vague. The use of language such as "Use of roughness elements such as woody debris jams *will be considered* to manage the post-construction roughness *to some degree*" and "*Design specifications will be developed* for hydraulic and geomorphologic circumstance for each stream type" indicates an incomplete plan. *All* details of the protection plan need to be clearly articulated; it cannot be assumed that issues will be resolved as they are encountered.

7. Stream grades. The post- and pre-mine grades are different in Figure 9 of the *Fish Protection Plan* document. Why, and how will this affect surface and subsurface hydrology, soil, moisture, and vegetation? Among other things, soil moisture plays a key role in the re-establishment and successional development of the riparian vegetation, which in turn controls the biological productivity of stream/riparian systems. Changing the grade will change the natural processes that affect natural chemical and physical patterns, affecting BMI, fish, and other species.

8. Reference reaches. There needs to be more extensive BMI sampling, including above (reference) and below (potentially impacted) the mining reach. Additionally, representative nearby stream systems not influenced by mining or other impacts need to be selected and sampled as reference sites for detecting eventual, potential mining impacts, and for monitoring rehabilitation efforts.

III. Conclusions

The individual studies reviewed for this report provide important biological information about the Chuitna system. My main concerns, however, have to do with critical information that is missing from the baseline reports and the unavoidable impacts to the Chuitna system, which are not addressed in the current plans. My concerns are summarized here.

1) By all accounts it appears it will be impossible to recreate the complex 3D network and interconnected underground channels of variously sorted sediments typically found below and lateral to streams, including streams like 2003. Flowpaths that influence aquatic productivity, and salmon spawning and egg development, which depend upon these hyporheic and groundwater networks, will be severed during the mining process. Recreating these highly complex and sorted networks and flowpaths in a fashion that reconnects them to the natural flowpaths of the intact, surrounding sediment veins will not be possible.

2) Compaction and settling over time of the refilled mined area will change the nature of surface flow in these areas, changing them from what they were previously into something unknown and impossible to predict.

3) Nothing is known about the actual food webs themselves, including what prey are important for the fish, where these prey are produced and delivered to the fishes, and when and where they are important. This needs to be understood in order for mining and reclamation plans to be developed that will maintain existing aquatic productivity.

4) General trophic connectivity throughout the watershed (upstream-downstream connections) is unknown but also needs to be understood for the same reasons as above.

5) The sampling completed to date has been inconsistent and insufficient. Multiple years of consistent sampling is needed to provide critical information on long-term annual variability. I recommend a *minimum* of 10 years of consistent and continuous aquatic biology and fisheries sampling before mining begins, to provide an estimate of the range of variability over a longer time period than the current data provide. This will also provide better statistical power when assessing impacts, including for the proposed BACI design recommended by LGL (LGL 2009).

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EXHIBIT 20

Chuitna Coal Mine baseline monitoring and restoration plan review

Executive Summary

The development of the Chuitna Coal Mine will lead to unavoidable impacts to the Chuitna system which are not addressed in the current plans. This scientific review found critical elements missing from the Chuitna system baseline aquatic reports. The failure to address food webs, trophic linkages, interactions among upstream-downstream, stream-riparian, stream-marine, and basin-wide linkages (below and above ground) severely undermines the ability of the mining plans to protect ecosystem function during mining and to restore it post-mining. Further, the lack of consistent, long-term sampling does not provide the needed estimate of annual biological variability or range of variability, and therefore does not provide a reference to which post-mining rehabilitation effects can be compared. Finally, recreating the structural complexity and interconnectivity of the below-ground sediment layers in the back-filled mine pit will be impossible, permanently and negatively affecting the natural flowpaths and hyporheic function (including natural upwelling and downwelling) upon which existing biological productivity and biocomplexity depend. While the individual studies reviewed for this report provide important biological information about the Chuitna system, the baseline reports are missing critical information. My specific concerns are summarized and reproduced here.

1) By all accounts it appears it will be impossible to recreate the complex 3D network and interconnected underground channels of variously sorted sediments typically found below and lateral to streams, including streams like 2003. Flowpaths that influence aquatic productivity, and salmon spawning and egg development, which depend upon these hyporheic and groundwater networks, will be severed during the mining process. Recreating these highly complex and sorted networks and flowpaths in a fashion that reconnects them to the natural flowpaths of the intact, surrounding sediment veins will not be possible.

2) Compaction and settling over time of the refilled mined area will change the nature of surface flow in these areas, changing them from what they were previously into something unknown and impossible to predict.

3) Nothing is known about the actual food webs themselves, including what prey are important for the fish, where these prey are produced and delivered to the fishes, and when and where they are important. This needs to be understood in order for mining and reclamation plans to be developed that will maintain existing aquatic productivity.

4) General trophic connectivity throughout the watershed (upstream-downstream connections) is unknown but also needs to be understood for the same reasons as above.

5) The sampling completed to date has been inconsistent and insufficient. Multiple years of consistent and continuous sampling is needed to provide critical information on long-term annual variability and to provide better statistical power when assessing impacts.

EXHIBIT 21

Main Identity

From: "Harold l. Rodgers" <hrodgers@gci.net>
To: <shaw.hahn@epa.gov>
Cc: "Tony Bickert" <tony.bickert@alaskastar.com>; <grammybeluga@aol.com>; <coho@ak.net>

Sent: Tuesday, August 08, 2006 10:09 AM
Subject: Chuitna Coal Project

Dear Ms. Hahn, (EPA Contact)

Allow me to introduce myself. This is Brenda R. Rodgers, owner of Three Mile Creek Sub Amended Lot 10 BLK 1. Both my husband and myself built a cabin on that property in 1982 after taking ten days and ten adults to just find the land mark stakes. Since then we have spent the last two weeks of months May through October at the cabin followed usually by weekly visits throughout the winter.

A matter of fact we just returned from over there where we found " no silence in our wilderness". The constant buzz of helicopters commencing at 6:00 a.m. and continuing throughout the day and evening drowned out the chatter of our visiting grand kids who wondered where the "silence had gone in our wilderness".

Husband Hal and I now wonder what is to become of our environment that has so richly provided us with an opportunity to fish, hunt, grow vegetables, and pick berries to supply us with food. Will that not only disappear, but so might we be burdened again with living in air and streams clogged by coal mining done with out attention to the gift God has given us of air, water and land? I'm 69. Hal is 73. He was raised as a coal miner's son in Kentucky. I was raised in St. Louis in an apartment heated by coal. We both do not remember coal for its warmth, but rather for the stifling air we breathed, the layer of black dust on the kitchen table and the destruction of wildlife/land/rivers.

Please let us know what is being done to temper so called progress.....for our health and the health of our wilderness environment.

Sincerely,

Brenda R Rodgers MPH, CHES
Harold L. Rodgers Maj Retired U.S. Army
18047 Sanctuary Drive
Eagle River, AK 99577
(907) 694-3392
(907) 583-2132 Beluga

8/8/2006

EXHIBIT 22

I grew up in Beluga From age 3 to 20, I visit now twice a year to help my dad commercial Fish and to experience the most wonderful magic place in the world. I still consider alaska to be my home and beluga is the center of it.

This coal mine, gigantic, deck and huge clear-cut of costal land For coal storage right directly on, over and around the fishing grounds used by me my father and generations of people who have lived and fished here like Frank grant who homesteaded the area in the 1920's is unbelievably horrible. the idea that I could return next summer to this monstrosity breaks my heart. I imagine what my grand father leonard mckay would have thought about all of this; he was the headmaster of the beluga power plant (a clean efficient producer of energy From natural gas abundant in the area) ? why would they do this to the chuitt and three mile beach when we wont even consider using the coal in our own country.

Alaska is pristine and beautiful we have an abundance of energy resourses not yet explored coal is not an option.

I live in a nice place in rural washington, The river I live next to is devastated by a large number of past mistakes pollutants/bad wildlife management. Alaska is one of Few places we have left to protect.

Please Stop the chuitna coal project.

Eric Jorgensen

P.O. box 123
maple Falls, wa
98266

RECEIVED

EXHIBIT 23

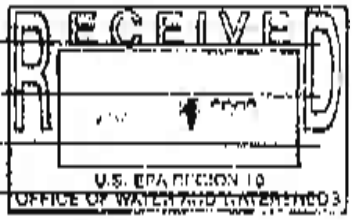
Scoping Comment Form

Chuitna Coal Project
Supplemental Environmental Impact Statement

If you have any issues or concerns that you would like EPA to consider in the Chuitna Coal Project Supplemental Environmental Impact Statement (SEIS), please complete this comment sheet, fold it on the lines with the return address showing, tape it closed, attach postage, and drop it in the mail to us.

If you prefer, you may submit your comments by email to Shaw.Hanh@epa.gov or by fax to 206-553-0165. If you do not have any comments but would like to be on our mailing list and receive a copy of the Draft SEIS, please complete the contact information below, fill out the box on the back of this form, and mail it to us.

Comments attach



Please provide your contact information. If you would like to receive a copy of the Draft SEIS, fill in the box on the reverse side.

Name: Helen Kuetz Affiliation: _____

Mailing address: 2125 N. Old Glenn Hwy

City, State, Zip Code: Palmer, ALASKA 99645

E-mail: _____

Please hand in your completed comment sheet today to ensure your input is considered, or if you would like to mail your comments, please use the address on the reverse side by close of the public comment period, July 24, 2006. **Please note:** Individuals making comments may request confidentiality. If you wish to withhold your name or address from public review or from disclosure under the Freedom of Information Act, you must state this prominently at the beginning of your comments. Such requests will be honored to the extent allowed. The EPA will not, however, consider anonymous comments. All submissions from organizations or businesses, and from individuals identifying themselves as representatives or officials of organizations or businesses, will be available for public inspection in their entirety.

Thank you for your interest and participation!

Fold comment form so that return address is visible; staple where indicated above.

Fold 2

Helen Kurtz
2125 N. Old Glenn Hwy
Palmer, AK 99645

Ms. Hanh Shaw
Chuitna Project Manager
U.S. Environmental Protection Agency
1200 Sixth Avenue, OWW-130
Seattle, WA 98101

Fold 1

Chuitna Coal Project SEIS Mailing List

To have your name added or removed from our mailing list for this project, please check the appropriate box and return this response sheet to EPA. Be sure to fill out the contact information on the reverse side. If you do not ask us to remove your name from our mailing list, we will send you future SEIS-related announcements.

☑ Yes, add my name to the mailing list to receive future information.

☐ No, please remove my name from the mailing list.

Sign up to receive the Draft SEIS

To receive the Draft SEIS, check one of the boxes below to tell us which format you would like.

Send me the Draft SEIS in the following format:

☐ CD-rom ☐ Printed Copy ☐ Printed Executive Summary

Printed copies of the Draft SEIS will be available at your local library or tribal office. The SEIS will also be posted on the Project Web site at www.ChuitnaSEIS.com.

HELEN KURTZ
2125 N. OLD GLENN HWY.
PALMER, ALASKA 99645
(907) 745-2232

1. This project will put a trestle 2 miles into Cook Inlet, which will cause sand bars to pile up and will force our salmon stocks out further into the inlet. This will destroy the commercial fishermen's livelihood. We know this will happen because the dock at Tyonek changed the salmon by pushing the salmon further into the inlet and we saw decline in catches. The fishermen have to set their nets out further for their catch, also jeopardizing their safety.
2. Sport fishermen will face the same problems when salt water fishing. As they have less experience with the Cook Inlet tides their safety will be more of an issue.
3. The impact of the loading facilities on salmon rearing and escapement should be done by the appropriate agencies. The local fishermen know that many salmon migrate along the shore and very narrow band of depths. Salmon spawn in the rivers and stream of Three-Mile Creek to the Big and Little Susitna Rivers pass by the proposed loading facilities. The appropriate agencies need to develop a baseline data, including solar counts, of the salmon movements around the facilities and require additional yearly surveys to access whether this structure is interfering with salmon movements and order corrective measures if it does.
4. There is little baseline data to provide and early warning if acidic or toxic runoff affects salmon and wild life. An independent third party should conduct baseline surveys of the quantity distribution and health of all fish and wildlife in the area and a yearly independent report should be issued assessing current conditions against these baselines. These annual reports should be distributed to all of the areas residents and landowners.
5. Local residents often use Cotton Wood and Three-Mile beaches as well as areas further north, (Chuit River to the Big Susitna) for recreation. Testing and monitoring of pollutants as they are directly affected by the winds from the mine and dock areas, as well as the tides of the inlet carrying pollutants up inlet.
6. The massive construction at both the mine and the loading dock will disrupt the local animal population. The human population increase to the area, with hundreds of workers, will additionally affect the hunting for the local residents. The local residents rely on moose and game bird populations. With little infrastructure to access animal movements away from these areas what is being done to address the loss of hunting?

7. The roads in the whole Beluga area are basic dirt roads. With the increase usage construction and of hundreds of workers traveling the roads, what will be done to upgrade and maintain these roads?
8. What is being done to address road dust pollution from all of the road traffic? Most local residents have subsistence gardens and road dust can affect these gardens as well as personal health.
9. Coal dust chemicals will escape and cover a huge area Coal dust WILL become airborne as well as filter down off of the conveyer belt. The proposed catch system will hardly be enough to address ground contamination. Coal dust will be produced all along the proposed corridor; at the mine, loading the conveyer belt, conveyer belt movement over land, unloading the conveyer belt, storage at the dock, loading the conveyer belt to the ships. Trace minerals and other pollutants will be a part of this pollution and must be addressed. Best Available Technology must be used all along the project for all pollutants. Independent experts should verify BAT requirements are being used. This should be done seasonally as different seasons will affect the accumulation of different pollutants.
10. As there are both pristine wild areas (the site of the mine being one), and permanent residents in the area, everything must have exhaustive dust control measures. The EPA should use BAT requirements for dust control for all phases of transport.
11. Any water produced from the mine or its operations should have BAT requirements to the emissions of water.
12. Are NPDES permits issued for each source of wastewater? The trace amounts of some highly undesirable pollutants will build up. How will the NPDES permitted source be independently monitored and inspected to make sure the coal company is operating within these guidelines?
13. The EPA should require that all reclamation and ground water control should follow BAT requirements. Even the modern mines currently run have water standard problems that will affect the area for generations.
14. With all of the pollution problems at Red Dog Mine (yes, we know that is lead not coal), we know regulatory agencies are deficient in controlling pollution, especially long term. Mercury, a powerful neurotoxin, is often found in coal deposits. Concentrations of heavy metals will take centuries to return to normal levels. Fishing and hunting for consumption has been prohibited because of the contamination. The EPA should control and monitor it's own regulations and other requirements and not cede any control to state agencies, including the DEC. All local permanent residents depend on the fish and game as food stocks.

15. Given the overlapping regulations and requirements, the coast Guard and the EPA should enter into a Best Available Technology requirement for pollution control, including dust mitigation, water discharge and emissions for it is imperative these agencies work together to insure BAT over the complete loading area and not have anything slip through the cracks of responsibility. The coast Guard and the EPA should enter into a Memorandum of Understanding for oversight.
16. All residents of the area believe the public notification process for the project and especially the supplemental EIS have been deficient. All agencies involved in the notification process should present a list of any attempts to notify residents of the area and other concerned parties, encourage public involvement and discussion. They should also perform an independent investigation agency, regulatory and procedural requirement for public comment and issue a report on whether any of these requirements for citizen and landowner involvement were done.
17. The Surface Mining Control and Reclamation Act is focused on contour control, acidity control and land restoration. The history is that it has ameliorated much of the damage strip mines like the Chuitna cause, but has not fully prevented impact. The existing federal regulations were oriented for the lower 48 states, which have little resemblance to the Chuitna. Therefore, the EPA should create a working group composed of regulators, industry, and local residents to develop additional requirements.
18. What are the concerns and objections of the Trustees of the Mental Health Lands with regard to this project?
19. What other sites for the loading dock have been discussed and what is the outcome of those discussions? Has Granite Point specifically been evaluated?

20. We can see the devastation of this area that took place in the 1932 earth tremor. This is a major fault line where land landing is. what percentage of safe-guards are figured into this? And what are they? (fire protection, movements of the trestle, contamination, and etc.) also, I do not believe the bonding of this project is high enough for construction on a MAJOR FAULT AREA.

EXHIBIT 24

To: Ms. Hanh Shaw,
Re: Scoping Comment Form,
Chuitna Coal Project.
July, 24, 2006.

During the summer months, I live at Beluga on a bluff overlooking Cook Inlet, often observing, rather ironically, a threatened local species swimming by in their hungry, playful pods during July, feasting on salmon and other marine nutrients. Our combined future may be at stake here – at least our hitherto lifestyle, working and recreating, ingesting clean air and water, and sharing the simple joys of a pristine natural environment.

Despite the rather general nature of this argument, I write my letter with passion and conviction edged with a bitter sadness that a pristine wilderness area along the jeweled shoreline and hinterland of Alaska's western Cook Inlet region is under siege from a proposed strip mine development. Apparently, the locals here are facing a "cash-and-carry" company, licensed in Delaware, owned by Texans, and created for Korean energy needs. It would seem that Alaska doesn't even benefit, except from a percentage of the profits. And, at the end of the day, after the proposed construction phase, there will be a net loss of jobs in the area because of the impacts on the environment and the decline of tourism. We are trading a renewable economy for a non-renewable one! Such a lack of forethought and environmental integrity! Such blatant avarice! The pregnant, 2-minute pause of Chuitna Coal's project manager, when asked what we locals would gain from this, "spoke" volumes.

This little community, comprising year-round residents and lodge owners, fish and game guides, business owners and commercial set-netters, not to mention the seasonal tourists and Chugach Electric plant employees all enjoy a quality of life here, 40 miles west of Anchorage. At our 2 hour residents' meeting on Thursday, July 20 every single person emphasized his/her need for a continuation of the clean, pollutant-free environment so valued at Beluga. We were all very clear about one thing in particular: that the proposed Chuitna Coal Mine will poison the rivers and waterways, destroy the surrounding vegetation, introduce a distinctly nasty blackened character to the sandy beach, and will eventually compromise the health of its residents. Make no mistake: little good will come of this, as the cumulative effects of a coal mine will destroy the animals, fish and birdlife of the region; the beauty and natural wilderness of the area will be a thing of the past.

> "Everyone has the right to use the commons but never to injure its use and enjoyment by others." (Robert F. Kennedy Jr.: Crimes Against Nature.)

Thank Goodness I can pen my dissatisfaction to the EPA and receive a fair hearing.

Since 1994, I have accompanied my husband, a commercial set-netter of over 30 years to our summer residence and workplace. Every fishing day, I make my way to the beach, to a specified shore leased area with a sack lunch for the boat crew, and enjoy a beach stroll, scanning the sands for golden agates, prized for their polished translucence. I look

forward to my walks along the beach; there is something peaceful yet invigorating about being on a beach, in all weathers. During fall, we spend much of our time commercially fishing the coho salmon, icing down and packing them ready for shipment to the processors in Anchorage. Together, we have made a life, sharing the wild and rustic beauty with family and friends, many of whom have traveled from all parts of the globe – Australia and Europe – as well as various parts of the U.S., to savor this divine little niche we call "home". Our cabin commands spectacular views of bald eagles and shorebirds that compete for the choicest salmon in Three Mile Creek, moose families and the occasional bear along the water's edge, fox and coyote on an evening hunt, as well as the aforementioned Beluga whales making their way up the Inlet. Every visitor, without exception, has marveled at the unspoiled wilderness, from the green forests and wetlands to the clean streams and lakes, to the sharp clarity of Mt. Spurr, in all its detailed glacial splendor, rising to the west. The famous and prolific salmon streams are by far our best asset as we see it, attracting those with a rod and reel and the energy to dance with those sparkling silver beauties every summer. We all take it for granted that they are born in and return to crystal clear, pristine waters, and when feasting on them, always presume they are 100% free of pollutants, Alaska being so famous for its wild, organic salmon.

An Aussie friend of ours who visited some years ago regretted his fishing lay-over due east of here. "Why did I even bother with Canada when THIS was waiting for me!" he remarked emphatically. Tall tales or "yarns" have been told and re-told back in the place of my birth regarding the size, quantity, and quality of these premier salmon, now classified internationally as "super food" by clinical nutritionists. So, what is to become of their fabled quality, and indeed their population, when their spawning grounds are inevitably contaminated with the chemical effluent from the proposed Chuitna Coal Mine dumping a toxic compound of iron, manganese, and ammonia, among other chemicals, into feeder streams and their "mother", Chuitna? This major river itself can hardly deal with the torrents of water bombarding it from the mine, let alone the ensuing poisonous waste products. In short, the Chuitna, its little tributaries, and their life force- the salmon – will fall into history and legend.

Why kill off all this majesty for the sake of selling our "soul" (coal) to Korea? Shouldn't we be searching out alternatives to fossil fuels? Isn't the idea of responsible development to decrease the effects of global warming, not increase them? Apart from the ugly profiteering aspect, surely there are ethical principles at stake here which need to be addressed by the EPA.

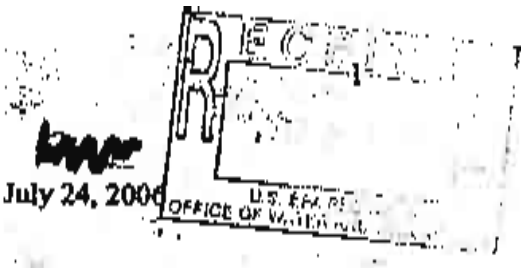
We do not want our beautiful environment destroyed by the ugliness of a polluting strip coal mine and all that goes with it. We all want to continue enjoying Beluga's four seasons, free from air pollutants and the visual monstrosities that are planned along the beach. We want the salmon to continue their life-cycle unhindered and unsullied. We want the waterways pristine the way Nature intended. We want to remain here, living healthy, unthreatened lives. The looming specter of a coal mine in our vicinity will, with little doubt, threaten to destroy everything we have here.

Lyn Jorgensen.

EXHIBIT 25

Scoping Comment Form,
Chuitna Coal Project

July 24, 2006



Ms Hanh Shaw,

Thank you for extending the scoping comment period, which was difficult to give my full attention to during the salmon season. Please replace my previous letter with this one, now that I have had a chance to read the previous EIS and listen to my neighbors' comments and concerns about the effects of this proposed strip mine. My name is Terry Jorgensen and I have worked at Beluga since 1974 and as a full time commercial salmon fisherman since 1981. Because the proposed Chuitna Coal dock is to be built on top of my fishing nets I know of no party more directly impacted by this venture. Commercial fishing and this present dock proposal as presented in the Supplemental E.I.S., dated June of 2006, are mutually exclusive and will effectively put me out of business along with the other four fishing sites on Three Mile Beach.

I hold a valid 10 year State of Alaska Department of Natural Resources Shore Fishery Lease that runs from May 31, 2005-May 31, 2015. This lease starts one mile from the Chuitna River and extends 2,100 feet to the north east. It extends seaward up to 725 feet and is directly in conflict with the unreasonable bulk dock proposal. The conveyor coal belt and huge 300 acre coal storage facility would also be directly overhead, depositing coal dust in my lungs, boat, beach, and into the water where I directly fish. This direct impact at tidewater is serious but not as important as the long term cumulative impact this proposal would have in permanently destroying the majority of all salmon resources in the Chuitna River and clean living environment for the residents who live in Beluga.. After attending a Beluga community meeting where not one local citizen favored this development, and after reading the entire original EIS, I completely oppose this development. I appreciate the opportunity during the scoping process to place in the public record and identify the issues I feel are relevant in creating sound public policy. The major issues are listed below with the kinds of data I feel are necessary to help the public understand the extent of the profound long term consequences this mine will make in a pristine wilderness. In conclusion I have listed alternatives that I feel are in the public interest if the mine is permitted.

:

Environmental Concerns

Dewatering and strip mine effluent impacts on freshwater salmon environments that may destroy the integrity of the Chuitna River drainage

As it states on page 5-13 of the draft EIS: " Impacts to the ground-water regime as a result of mining operations would be substantial and would affect recharge and discharge relationships: quantity, and quality and direction of ground-water flows: and quantity and quality of surface water."

In the proposed mining area there will be four streams that will be immediately and permanently impacted by coal pollution, changes in hydrology, and resulting loss of salmon habitat. The most important aspect of each is that they will drain directly into the Chuitna River, permanently altering the most significant subsistence, sport and

commercial fishing river in this entire area. The Chuitna is 17 miles long and at a point 0.6 miles below stream 2002 (Lone Creek) it leaves a steep walled valley and forms riffle sections interspersed with deep pools. It was rated in the EIS as "very high in habitat value for Chinook and high for all other species because of its combined function as a migratory pathway, spawning area, rearing area, and excellent sport fishing water." This most productive fish habitat in the entire Beluga area is directly below the massive dewatering of 7,245,000 gallons of water per day from the strip mine that will be allowed into tributaries 2002-4 and then dumped into the main stem of the Chuitna.

The following is specific information of each stream listed as a "significant Chuitna River tributary that drains a portion of the mine area" EIS p.4-37 -38. It is important to note the irreparable long term loss of habitat when all area leases are mined and 90% of these streams are altered, along with Chuitna Creek, another major tributary to the system.

Stream 2002 (Lone Creek)
1 Flow contribution to Chuitna River. (ERT 1985c) 15%.
2 Number of tributaries within creek system (EIS Figure 4-5) 9
2 Mean annual flow m/sec (cfs) 1.36 (48.40) Table 4-6 ERT 1984a
3 Daily anticipated mine runoff flow requested in EPA Permit is 725,000gpd
4 Rearing composition of Juvenile Salmonids (ERT1985 a,b.) See figure 4-7 Chinook, Coho, Pink
5 Salmon escapement percent to the Chuitna River system 1982-4. See table 4-12 (note that counts were conducted in different portions of stream from 1982-4)

Chinook 5-7%
Coho 9-23%
Pink 3%

6 Assigned lower reach habitat value EIS Table 4-13(ERT a,b,c.) – Very High

Stream 2003
1 Flow contribution (ERT 1985c) 7%
2 Number of tributaries within creek system 7
2 Mean annual flow m/sec (cfs) 0.71 (25.37)
3 Daily anticipated mine runoff flow requested by Chuitna Coal in EPA Permit 3,140,000
4 Rearing composition of juvenile salmonids (ERT 1985a,b) Figure 4-7 Chinook, Coho, Pink
5 Salmon escapement percent to the Chuitna River system

Chinook 1-14%
Coho 5-9%
Pink 4% to N.E.

6 Assigned lower reach habitat value EIS Table 4-13 (ERT a,b,c) –Very High

Stream 2004
1 Flow contribution (ERT 1985c) 10%
2 Number of tributaries within creek system 12
2 Mean annual flow m/sec (cfs) 0.09 (3.33)

3Daily anticipated mine runoff flow requested in EPA Permit 3,380,000
4 Rearing composition of Juvenile Salmonids (ERT1985a,b) Chinook, Coho, Pink
5 Salmon escapement percent to the Chuitna River system
Chinook 1-8%
Coho 3-9%
Pink 0-N.E.
6 Assigned lower reach habitat value EIS Table 4-13 (ERTa,b,c)–Very High

Fresh Water Issues Identified And Data Needed.

A. Maintaining proper stream flows

1. With massive dewatering from the strip mine what will be the short and long term impacts on streams 2002-04?

2. River flow will be measured monthly by lessee if permit is issued. Which specific streams and locations will be monitored?

3. The lessee will measure this flow if permit is issued. What provisions will be included as oversight by state or national agencies?

4. Will specific benchmark penalties be included if the Clean Water Act provisions are violated and the main stem of the Chuitna is polluted?

5. What is the anticipated percentage of decline from these dewatering and mine effluent volumes in relation to the entire Chuitna salmon and trout populations during this 25 year process?

6. Who will monitor this decline? At what intervals?
7. How much of the decision making of this proposed mine will the Department of Fish and Game, both sport and commercial fish, be granted in the determination of permitting and compliance?

8. What if any incentives will be included as provisions of the lease to protect the fresh water environment?

9. 90% of Chuitna runoff occurs during 10% of the year and creates the present fresh water habitat. How will continuous dewatering and mine effluent impact the depth of large pools necessary for salmon regeneration? Can a study on this be included in the final EIS?

10. As indicated in Table 5-7 minimum flow in the Chuitna immediately below the mouth of Lone Creek could be reduced by 17%. What will be the biological impact on eggs in spawning areas that need constant water flows?

11. Will minimum flow levels be established by the State of Alaska and be monitored?

12. The present course of stream 2003 will become extinct according to page 5-7 of the EIS. What levels of salmon restoration will the permit require?

13. Has any salmon stream ever been reclaimed in North America to its original state of production, utilizing specific coal permit provisions after being strip mined?

14. What are the specific bonding costs that would replace 25%, 50%, 75% or 100% of the loss of each of the following in the Chuitna watershed:
1. Subsistence food sources; 2. Sport fisheries; 3. Sport fishery, guiding, and lodging; 4. commercial fishing?

B. Surface and ground water quality in the Chuitna Drainage

If over 7,000,000 gallons of water are permitted to enter the Chuitna it is important to identify its present water composition and quality as compared to the mine effluent. While reading the old EIS, two issues standout. Firstly, how few wells were drilled in the mine area itself to establish a baseline from each coal and water column. Secondly, how varied different seams of coal in this formation are in their chemical composition.

Data reported by Scully et al. (1981), ERT (1984c), and Mauer and Toland (1984) indicate that the water quality of streams in the Chuitna River basin is consistently high throughout the year, which is typical in pristine areas of Alaska. Mineralization is low as indicated by relatively low conductivity levels, ranging up to 120 microns per centimeter at 25degrees C. Groundwater from the majority of streams is primarily from shallow aquifers with water quality similar to surface water. The exception is Lone Creek, located in the mining district. As the EIS States on page 4-33, "The water quality of the upper part of Lone Creek appears to be slightly affected by discharge of ground water originating from deeper aquifers." Because the proposed mine will be up to 300 feet deep and discharge water into the watershed it is important to look at the sampling of wells that penetrated the different coal seams that will be exposed. From data below for example if the mine dewaters from sub red 1 sand level it is 758% higher in minerals than the present Chuitna River water.

Mineralization as seen in conductivity levels at micramhos/cm @ 25 degrees C

Chuitna	Overburden	Blue	Red 3	Red 2	Red1	sub red1 sand
120	250	280	400	590	580	910
% of Chuitna	208	233	333	491	483	758

Elements measured in mg/l. Values are averages for each coal unit.

Sulfate ?	5.8	21.2	2.5	38.9	39.7	5.9
Iron ?	5.1	2.1	2.1	12.1	39.7	3.7
Manganese?	0.39	0.29	0.08	0.16	0.72	0.15
Zinc ?	0.23	0.41	0.22	2.24	2.51	3.39

Mercury (Not measured)
Arsenic (not measured)
Lead " "
Boron " "
Nickel " "

Chuitna Water

Ground water outside project boundary in the Chuitna River drainage "easily met drinking water standards" and trace elements were low. EIS P. 4-33.

Coal Seam Waters

Ground water in all coal units except springs exceeded drinking water standards for iron and manganese. Iron, zinc, and ammonia also exceeded the level critical for the preservation of freshwater aquatic life. Iron in the Red1 level for example exceeded the aquatic life standard by 39 times. On page 3 Permit No AK-004357-5 Effluent Limitations and Monitoring Requirements states the daily average of iron is 3.0 mg/l and daily max is 6.0 mg/l. In looking at iron concentrations from coal levels Red2, Red1, and sub Red1 Sand, how can any water be pumped without violating this permit? Concentrations of boron, nickel, and zinc are low, but have found to occasionally exceed recognized standards for aquatic life. If water is pumped from the coal level Red1, it is over 14 times as polluted as the Chuitna River at present for zinc.
"The water in the coal development lease area exhibits significant variation with depth or stratigraphic position of each hydrogeologic unit" EIS 4-33. The Chuitna River water was not included to compare with coal seams, yet if mineralization, iron, and zinc are an accurate measurement, pumping these coal seams to dewater the coal will create a toxic soup and sterilize the most productive stretches of the Chuitna.

pH is a measure of the level of activity of hydrogen ions in a solution, resulting in the acidic or basic quality of the solution. The pH range is from 0(acidic) to 14 (basic), with 7 being neutral. Stream organisms are adapted to certain pH ranges. Most natural rivers range from 6.5 to 8 units. Humans impact pH through mining activities and related leaching which makes the water more acidic. The present permitting process standards for pH are to remain between 6-9.

Issues Identified and Data Needed:

1. Will a base standard for present water quality be established for the Chuitna River and its tributaries and Three Mile Creek to be reviewed by the public in the final EIS?
2. What is the average conductivity level of the Chuitna in 2006?
3. What is the average monthly suspended solids level in the Chuitna?
4. What is the Chuitna average for:
    - Iron
    - Zinc
    - Ammonia
    - Manganese
    - Arsenic
    - Mercury
    - pH level
5. If element levels exceed aquatic life standards what water treatment will they undergo to meet these standards for each element before over 7,000,000 gallons per day are being discharged into the Chuitna by the proposed coal mine?
6. Will these standards be applied during runoff in the spring when the majority of leaching will occur?
7. Because of the toxic composition of coal seam water how many wells will be drilled for the 2006 EIS to delineate water quality for the mine permit?
8. Why were coal leaching tests not included in EIS?
9. Exactly what levels of Aqueous Hydrocarbons are expected to be released into the onshore and offshore watershed.
10. How will the coal storage waste water at ____gpd be treated to meet the EPA guidelines for discharge?
11. Will waste water be tested at the edge of mixing zones or at the source of outflow, according to the permit requirements

C. Surface and Groundwater Quality in Three Mile Creek

Three Mile Creek is a small stream that drains a large, fairly flat area immediately south of the proposed mine and will be directly impacted from air and water contamination. It has three main tributaries. Because I have also hunted moose in this area for 30 years, I have been amazed at the ability of Sockeye to travel so far up such small streams. I have frequently observed them north of the old Pan Am air strip spawning in September. This stream also has runs of Coho that can be so abundant one cannot see the gravel in major spawning holes. My summer home is situated on a bluff overlooking this stream and it is hard to miss both runs which vary from year to year. The sockeye fry rear in the Three

Mile Lake system and Coho fry utilize these small tributaries to rear before returning to Cook Inlet.

Issues Identified and Data Needed

1. How will the water quality levels be monitored in this system to compare with the EIS studies?
2. What is the anticipated coal dust accumulation to this drainage when a prevailing north wind in the fall and winter meets an exposed conveyor belt?
3. Will any mine flow be diverted into this drainage in the 25 year lease proposal?
4. What water volume can it handle before degrading this rearing area?
5. How will fish production that has always been natural be monitored?
6. What if any specific Three Mile water quality responsibilities will be written into the lease for both the leasor and leasee?

D. Coal Air Pollution

From Appendix E Air Quality Emissions as calculated by the State of Alaska Department of Environmental Conservation, we can clearly see the impact of this mine in the mine area, conveyor system, and port loading facility:

Mine Area full production

Primary coal crushing	= 1.2 ton year	
Secondary coal crushing	= 3.6	
Wind Erosion Mine	= 35.9 tons yr	
Coal screening	= 6.0	
Mine Stockpile	= 20.5	4lb per/acre-hr control 50% with watering
YEARLY TOTALS	=67.2	

Conveyor System full production 2 ft belt

8.4ton/year 8.13lbs/acre-hr with 2foot wide conveyor belt. (note:. the latest proposal is for a 7 foot wide belt. 90% control with hood.)

Port Facility at full production

Mine service facilities Area	10.0 ton year. hourly	2.3lbs
Coal pile Port Facility	218.1 ton year. hourly	49.8lb
Wind Erosion Port Facility	11.9 ton year. hourly	2.7lb
Totals	240	54.8

Since I commercially fish directly south of the proposed loading and storage facility, I can not foresee a continuation of any fishing by me or by anyone else in the area without severe health consequences, if this permit is granted. My boats, nets and equipment, and base of operations I lease from Ward Grant, where we mend gear and hold salmon while awaiting transport, is directly east of this massive storage facility. The entire acreage will be blackened within a few years with typical Southwest and Southeast wind patterns. Because the Three Mile Creek subdivision is located only 1-

3 miles to the East, the same will be true for this entire area over time. Every neighbor at our July 20th meeting felt property values will drop drastically as a result.

Issues and Data needed for Air Pollution

1 Can expected coal dust accumulation be charted for local residents for our comments on the final EIS?

2 Why is a shore-based stockpile necessary in an ocean beach area known for its particularly strong winds?

3 What are the anticipated health impacts to Tyonek and Beluga residents?

4 What are the socioeconomic costs associated with a 100% enclosed conveyor system?

5 What are the socioeconomic costs in moving the coal facility to Granite Point?

Commercial fishing on Three Mile Beach

In reading through the previous EIS from 1990 I found an almost total lack of information on commercial fishing impacts with this coal proposal. We the commercial fishers, or the Northern District Setnetters Association for that matter, were never contacted until after my testimony at the Public hearings on the mine permits held in December, 1986. The same has been true in 2006, when we discovered on July 12 that we had a mere 12 days in which to comment. In looking over the document, bears, moose appear to hold a much greater value than the fisherman working directly near the terminal for three to four fishing months per year. We were listed as a "possible conflict" by the Chuitna Coal EIS, yet no information was presented to Government review agencies to explain it. Surely, good decisions are a result of good information. I hope the following may aid in this effort.

Fishing Seasons

The King salmon season, by regulation, begins on the last Monday in May and extends for three 12 hour (7am-7pm) Monday periods. The Sockeye, Coho, Chum, and Pink season begins the first Monday or Thursday after June 25 and continues until the end of the calendar year for 12 hour periods, Mondays and Thursdays, unless closed by regulation. Additional days may be granted by regulation according to the Cook Inlet Commercial Salmon Management Plan. For specifics contact the Alaska Department of Fish and Game at 262-9611. Jeff Fox is the regional commercial fishing manager.

Salmon Production

The site I presently operate has been fished since the 1890's. It was so productive in those days that a cannery was constructed there, named Ladd, and another at the eastern

AK Pebble Debby - ADN 9

end of the beach, at Three Mile Creek. Since Statehood, tenders have taken this renewable salmon resource to Anchorage, Kenai or Homer to be processed. This site is annually one of the three most productive salmon fisheries in the Northern District of Cook Inlet. It is presently owned by the Grant Family, who have invested their lives in commercial fishing on Three Mile Beach. I have personally caught over 90,000lbs of salmon twice since 1980, fishing these locations. I have fished almost every opening allowed since 1980. Today, the fish are flown to Anchorage for processing, to be shipped out to specialty markets or sold for local consumption. Last summer, a chef from Tulsa, OK. visited, personally inspected the operation and has since purchased thousands of pounds of this product. He even created a video on the advantages of wild pure Alaskan salmon, filmed at the exact location of this proposed dock. (DVD available upon request.) Salmon Prices are up 100% this year after a decline since 1990, and the market remains strong if we are allowed to continue fishing. This socioeconomic factor was not even mentioned in the EIS document for this mine proposal. We hire from the local community to commercially fish, and engage the services of Spernak Airways and Great Northern Air to fly the fish to Anchorage. Fisherman's Express will attest that we are one of the most prolific producers of salmon for them in the Northern District. (See Alaska Journal of Commerce, June 2006, or contact Cade Smith at 884-4976. For a 30 year review contact Tim Schmidt at Seward Fisheries in Homer at 235-8107.)

Present net locations and fishing practice.
During the King Salmon season only one net may be used, and based on the tide, we fish it in all three locations. During low tide, I typically fish directly off the present landing; however, should this proposal come to fruition, fishing at low tide would be rendered impossible in the vicinity of a bulkhead or pile type dock.

During the "smelt fish" season, shore nets from the high water mark are fished on both ends of the lease and the deep net is fished directly at ground zero of the proposed bulkhead dock. . This deep net can only be fished effectively around low water and for a few hours due to the velocity of the tides. If not removed at high water the deep net would be sucked under and extremely difficult to remove on the ebb. All nets are 220 feet long. The deep net is typically set 220 feet from mean high tide, and two railroad car wheels anchor it from 550 feet off shore. Shore nets are most productive on the flood, but are fished all 12 hours because the velocity of the current is lessened by shallow water near the beach. The net location near the Chuitna River is in water 4-6 feet deeper than the remaining lease, and at low tide one can easily see this gradual shallowing of water extending to Three Mile Creek, and then to an extreme degree all the way to the Beluga River.

Dock's Impact on Fishing Practices
The proposed dock, extending 800 ft offshore, would completely eliminate the traditional movement of salmon stocks down Three Mile Beach by creating new currents and massive silt and gravel deposits. This monstrosity will impede the constant movement of gravel and silt from a typical southwest wind pattern and would easily form huge deposits on either side of the dock. The gravel moves in a W pattern and is finally deposited on the sand bar of Three Mile Creek. This pattern can be seen weekly as Bob Freeman,

owner of Three Mile Creek Services, bulldozes up gravel for barges to land. The depression his equipment creates is completely filled in after a typical afternoon Southwest wind.
Small deposits of coal is another way to observe this process. It tends to come down the Chuitna River in the winter in small amounts. After any Southwest wind, a small black line extends all the way to Three Mile Creek one and one-half miles from the proposed dock. If tides are in a receding pattern, three or four very small coal lines may be formed. With an uncovered coal conveyer and unnecessary, ugly coal stock pile, just how much would be deposited in Three Mile Creek? How will it impact returning salmon? How would the surplus coal deposits be cleaned up from the beach? And just who will be monitoring the resultant pollution of Cook Inlet? Any reasonable opportunity to fish this lease will be lost because it is simply not possible to fish 1,000 feet off shore in tides that average a 30 foot fluctuation in 6 hours. During the past 30 years, barges have been unloaded on a regular basis leased by Chugach Electric and numerous oil companies. Unlike the current coal interests, Beluga commercial entities in the past have worked together, and they are in constant contact with me to ensure they will not impact commercial fishing. There is no noticeable environmental impact in these barge-landing practices, and as such, must be maintained with this proposed coal mining process. Note also that this form of beach landing design was included in the original proposal in 1990 and was subsequently rejected.

Trestle Facility

In looking at the June 2006 Supplemental Environmental Impact Statement and the original EIS of 1990 I find that they are incomplete. How can a 10,000 ft facility be constructed without impacting the migration of salmon smolt, adult salmon, commercial fishing, and Beluga Whales? What electric current is necessary to prevent corrosion created by passing salt water. What impacts will ship wakes have on Three Mile Beach coastline erosion.

Shore Fishery Lease

In re-reading my Shore Fishery Lease issued from D&R Division of Mining, Land and Water and the Alaska Constitution, it is clear in my mind the issue is what are my *Valid Existing Rights* under section 12, as related to *Concurrent Use* in section 15 .
My lease is subject to "reasonable concurrent uses" as provided under Article VII, Section 8 of the Alaskan Constitution. In looking at Stare Decisis, what former Alaskan precedent relates to "reasonable current use" defining this constitutional issue? Both parties may have a property right if a lease is issued on top of mine, yet how could this process be reasonable if my opportunity to continue past fishing practices is in any way mitigated?
?

Marine Pollution

As I type this Scoping Comment Form, I can observe a southeast wind blowing at about 40 mph. These are locally termed, "Turnagain" winds by the locals, named for their source, Turnagain Arm. Historically, however, these winds were called "the cannon",

which necessitated steam vessels to be docked at Tyonek because of the dangerous seas. This summer, the strongest "Turnagain" has been 74 mph and lasted for three days during June. In 1989, the leaves were blown off my birch trees, and according to the USWS, it blew 114mph that day. I am unable to fish in these winds, even at 25mph, because they hit the beach head-on. They are not to be taken 'lightly', and should be included in the EIS created. In 1989, Phillips Petroleum had a supply vessel rupture a fuel tank at their gas platform during a "Turnagain" storm. I called, and they admitted they had "lost" some fuel. I told them where to find it because I had been standing on the beach with diesel oozing out of the sand. The smell was overwhelming even in the wind. Luckily we were not fishing at the time. They promised me that a spill response on shore unit would be created especially to protect the Chuitna River. That was the last time I heard from them.

This EIS needs data not only on wind speed but information on docking fuel vessels, that because of tidal movement, will be directly sideways to these southeast winds. Data also needs to be collected on Inlet salinity. The Northern District is atypical, in that considerable volumes of fresh water from the Susitna, Little Susitna, Lewis, Theodore, and Beluga Rivers flow on the top water column along Three Mile Beach. This creates unique shears and salinity. Data is available that clearly shows gradients from normal seawater at Moose Point, directly across the Inlet, to limited salinity at Three Mile Beach. Fresh water waves are steeper and create unique rips especially in SE winds that push more fresh water landward. Any dock or coal loading facility must include data on these potential problems.

Non economic Factors

How much is a 30 year lifestyle worth? How can I define the intrinsic value of watching my kids grow up helping me work and play on Three Mile Beach? What are the three decades of photos worth on my cabin wall representing a generation of friends and family? At the last coal public lease hearing, during my testimony, I became emotional when I saw its potential impact on our way of life. The coal interest comments were so bottom line and many environmental representatives were speaking from studies that seemed so technical, so devoid of the human factor. How do you place a value working with fishermen like Frank, Marilyn and Ward Grant, Rick Volk, Tom Hasbrouck, Ray Rockie, and my wife? I appreciated the time taken by the chairman at this hearing in asking follow up questions and including comments that make me realize how all Alaskans share places that are unique. Mine happens to be near a beach, acting as a steward of unique common property resources. I tried to apologize to my now-grown kids a few years ago for, in effect, "stealing away" their childhood as they spent every summer of their early growing years at Beluga. Both made me understand how they would not trade their experience for any other memories. Defining herself in a Fullbright application a couple of years ago, my daughter opened her letter with: "This may seem strange but I grew up in hip boots helping my dad as a setnetter". Picking blueberries within close proximity to bear habitat does build character, common sense and self reliance, yet how can you place a value on these lifestyle choices that are made by the residents of Beluga? Right now, the reality is I awaken to the jarring whirr of a fleet of helicopters laden with coal employees heading out to the fields every morning at 7am,

and realize, sadly, that life has already been altered. (Incidentally, how many shareholders in Pac Rim Coal would enjoy a lifestyle near the new conveyor belt?) When commercial fish prices collapsed after the Exxon Valdez oil spill, people asked me why I continue to fish. I answer then as I do now: How can I explain the palpable stress relief and peace of mind it offers; beachcombing walks and cook-outs on the sand with my wife; or living in a place where passing "beater" truck owners wave to each other and greet each other at the local landfill? This year, for example, during King salmon season, my overloaded truck suffered a blowout on my way to load salmon onto a waiting plane. An employee with the local construction and service company, on his way to a job, immediately stopped, re-loaded the entire, huge catch in his truck and took me post haste to meet the plane. His assistance was invaluable to me at the time as I was required to be back on the beach at my net. Interestingly, 5 different trucks stopped along the road to lend a hand as I was being assisted. This is a valuable lesson in humanity and more particularly, the mutual respect folks demonstrate in this small community.
I hope these non economic factors will be weighed as you reach your decision and represent the public interest.

Conclusion

In the early 1980's I tried burning coal to stay warm on the beach and in my cabin. I gave up for one major reason: it's so dirty. While it gives off heat, coal does so at a price I am not willing to pay. I feel as a young struggling state in the 1960's, we Alaskans granted these coal leases without any consideration of the long term consequences to such a pristine environment. At this time, Alaska had barely enough revenue to survive as a new state. In those days, it was "cash and carry". For enough "cash" you can "carry" off Alaskan resources. We all should recognize that the days of Kennicott copper and Swanson River oil have been replaced by the constitutional principle that Alaskan resources are to be utilized for the maximum benefit of all Alaskans. We should, therefore, commit as a people, to buy back this property right we granted to a Delaware company controlled by Texans for the benefit of Asian hydrocarbon needs. The Environmental Protection Agency in concert with other state and federal agencies will now take these scoping letters to determine what will be required in the final environmental impact statement. The process has been overwhelming, especially when the original EIS is so outdated and incomplete. I hope you will take all of the letters from local citizens seriously, as guardians of the public commons. For the sake of this area, it must be done right if the permits are granted. My neighbors, like myself, were unanimous in opposing this mine at an open community meeting, attended by 20 locals last week.
Thank you for your time, and I suggest the following for your consideration:

Suggested Alternatives

1. Eliminate proposed 800 foot solid bulkhead dock. Use existing barge landings and continue the process that has worked for 30 years at Beluga. Factor SE winds into lease agreement.

2. Locate Ladd coal storage as far from Cook Inlet as possible to limit pollution in Cook Inlet.

3. Water down, compact, and cover all coal at Ladd to limit dust in surrounding area. Limit all coal storage piles to 35 feet to avoid air pollution.

4. Eliminate all coal storage at Ladd.

5. Eliminate all mine runoff that flows directly into the Chuitna River drainage.

6. Require monthly non mine company monitoring of all area streams and air quality as well as noise pollution stations.

7. Eliminate mining from any coal seam or water column that is known to exceed the Clear Water Act standards during spring runoff.

8. Eliminate dewatering of coal to safeguard water resources and lower air pollution.

9. Require baseline testing of water table in entire mine area to understand future impacts.

10. Require lessee to create abatement and cleanup schedules for Cook Inlet Beaches, local roads, conveyor corridor, local properties, and mine dust accumulations near mine itself.

11. Limit construction in Cook Inlet during smolt migration, peak runs of salmon, and commercial salmon fishing days.

12. Ban all barge landings and coal loadings of vessels during commercial fishing openings. Presently: one in May; two in June; and each Monday and Thursday throughout July and August.

13. Cover entire coal belt to limit pollution.

14. Require an on site marine pollution response plan to protect river estuaries and Cook Inlet beaches

15. Build loading facility at Granite Point to mitigate SE wind pollution problems.

16. Use existing dock at Tyonek with improvements for all freight handling.

17. Utilize the existing Tyonek dock for all fuel transfers. Fishermen and tender operators have known for generations that to escape a SE wave you need to get west of the Chuitna River delta to avoid this.

18. Utilize as many natural gas burning vehicles in the coal vehicle fleet and electric shovels in the mine.

19. Add to the EIS Document the socioeconomic and non economic considerations for all people impacted in Tyonek and Beluga.

20. Identify what existing rights are included in a shore fishery lease. Identify the relationship of reasonable concurrent use in balancing economic and non economic considerations.

21. Create an incentive for the Lessee to adhere to terms of lease by requiring five and ten year terms for renewal. These should be based on specific terms representing the public interest in the lease.

21. Ensure sufficient bonding in the following foreseeable affected areas: commercial fisheries, sports fisheries, subsistence, guides and lodge owners, Chuitna watershed restoration, Chuitna salmon restoration, Beluga wildlife restoration, mine soil reclamation, Beluga wetland restoration, coal dust pollution clean-up, local (Beluga and Tyonek) health of the population, to allow mitigation from the local impacts of coal strip mining.

Contact Information:
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907-583-2662

Add my name to all mailing lists. Please send a printed copy of the draft SEIS and the executive summary.

EXHIBIT 26

STATE OF ALASKA

[redacted]

August 7, 2006

Hanh Shaw
Chuitna Project Manager
U.S. Environmental Protection Agency
1200 Sixth Avenue, OWW-130
Seattle, WA 98101

Re: Chuitna Coal Project SEIS Scoping Comments

The Alaska Department of Natural Resource, Office of Permitting and Project Management (OPMP), has conducted a review of the Draft Scoping Document prepared by the U.S. Environmental Protection Agency (EPA) for the Chuitna Coal Project Supplemental Environmental Impact Statement (SEIS). We are submitting the following comments covering the mine and associated support facilities such as an access road and utility corridor, coal transport conveyor, personnel housing, airstrip facility, and export terminal.

GENERAL COMMENTS

1. Baseline Studies and Impact Assessment: The fish and wildlife species in the area support substantial commercial, subsistence, and sport uses. We anticipate that the identification of potential impacts to the fish and wildlife resources, their habitats, and human use of these resources within the area of proposed mine facilities will be a substantial component of the SEIS, as well as the operation and reclamation plans prepared by the applicant. It is important that the baseline studies be of sufficient detail to allow for the identification and avoidance of potential impacts to habitats or uses. The SEIS timeline should allow for the collection of additional data if existing data is determined to be inadequate.

The baseline data for surface hydrology, groundwater, and water chemistry will be important in assessing potential impacts to aquatic resources. These data should include water chemistry and physical characteristics that may be affected by development such as temperature for both surface and groundwater. Groundwater models should be used to generate dewatering discharge volumes in the project area. Sufficient hydrologic data should also be collected to describe the seasonal and long-term flow characteristics of the streams both within and downstream of the mine area boundary.

The assessment of potential impacts will require information on all project components, including the proposed locations of material borrow and disposal sites for the construction of

roads, parking pads, and other facilities requiring fill borrow material. All project components must be designed to avoid and minimize: (a) the disturbance of drainage systems; (b) changing the character, polluting, or silting of state waters; and (c) the disturbance of fish and wildlife resources.

2. Cumulative Effects: Project impact assessment must include analysis of cumulative impacts from the proposed project and other current and reasonably foreseeable future developments in the entire Cook Inlet region. The Draft Scoping Document solicits information on past, present, and future actions within a fifty mile radius of the proposed Chuitna Coal Project. We suggest that the EPA include in its cumulative impact assessment activities within the effected watersheds and Cook Inlet that may add to the collective significance of this project. Cook Inlet ship traffic will increase as a result of the proposed mine. The proposed Port of Anchorage expansion will also increase the size and number of ships traveling in Cook Inlet. The development of infrastructure for this project could lead to the future development of additional Beluga Coal projects. These and other projects in the Cook Inlet region will significantly increase the potential cumulative impacts of the Chuitna Coal Project.

3. Human Uses: All project components should also be designed to avoid and minimize impacts to access and existing uses such as set-netting, recreational fishing and hunting, and subsistence activities. Assessing the potential impacts of the various project components to human use of fish, wildlife, and plant resources involves a separate and distinct analysis from assessing impacts to the resources themselves. Information pertaining to the commercial, sport, and subsistence harvest of fish, wildlife, and wild plant resources, including harvest levels and harvest locations, should be collected to make this assessment possible. It is our understanding that the project applicant is working directly with the ADF&G Subsistence Division to provide data on subsistence uses in the project area.

Every effort must be made to maintain public access to and along the beaches and other lands in public ownership within the project boundary. The economic impacts analysis should assess the potential loss of fisheries resources, habitats and sport angler dollars due to the development of the project.

4. Mitigation/Reclamation: The SEIS should also address how unavoidable habitat impacts from the development of project infrastructure will be mitigated. In order to adequately assess alternative mitigation options, a sufficient description of baseline conditions and the expected level of impact is needed along with an economic valuation of the impacted resources. The updated economic analysis may be used to develop potential value-based mitigation requirements. In addition, plans for reclamation must be of sufficient detail to evaluate the potential for success, and should include timelines to fully realize restored habitat and ecological functions. One of the key project documents addressing habitat impact mitigation will be the reclamation plan for the project as required under the Alaska Surface Coal Mining Control and Reclamation Act.

5. Spill/Contingency Plans: The SEIS should address the potential for spills, and the possible impacts to Cook Inlet and area streams, lakes, and wetlands from spilled material including but not limited to fuel and other petroleum products, coal, and untreated wastewater. The SEIS

should evaluate the adequacy of the applicant's contingency plans for spill response to protect the environment in the project area.

6. Water Quality: The construction and operation of all components of the proposed Chuitna Coal Mine will result in wastewater discharges to the Chuitna River drainage and to Cook Inlet. Wastewater and surface runoff from all project components should meet or exceed the State of Alaska water quality standards. The SEIS should address disposal alternatives for all wastewater discharges and evaluate the potential impacts that wastewater dicharges may have to the freshwater and marine resources in the entire project area.

Coal Mine and Mine Facilities

7. Habitat Loss: There are important habitats and valuable fish stocks in the project area. Middle Creek (Stream No. 247-20-10010-2030) and three unnamed tributaries of Middle Creek lie within the area proposed for mining. Middle Creek and its tributaries have been specified pursuant to AS 41.14.870(a) as being important to the spawning, rearing, or migration of anadromous fish. These four streams provide five and one-third (5.3) miles of known rearing habitat for coho salmon, three (3) miles of known spawning habitat for coho and pink salmon, and two and one-quarter (2.25) miles of known rearing habitat for chinook salmon. These streams also support rainbow trout and Dolly Varden char. In addition to Middle Creek and its tributaries, many other streams and lakes in the project area may support populations of resident or anadromous fish. For any unavoidable aquatic habitat losses, mitigation in the form of habitat creation in another portion of Middle Creek should be considered in addition to reclamation and restoration of the creeks at mine closure.

Middle Creek is a tributary of the Chuitna River. The Chuitna River provides the greatest opportunity for harvest of chinook salmon among West Cook Inlet rivers and also supports valuable coho and rainbow trout fisheries. Loss of aquatic habitat in the Chuitna River drainage may impact the commercial, sport and personal use fisheries in the region. The SEIS should examine the impacts of habitat loss in the mining area as it relates to the productivity of the entire Chuitna River system. Construction activities and work within these waterbodies will require Fish Habitat Permits from OHMP provided plans and specifications have been developed that will ensure the proper protection of fish and game resources.

8. Dewatering: The applicant proposes to dewater the overburden and interburden material to facilitate coal extraction. The SEIS should evaluate the potential impacts of dewatering to freshwater aquatic resources in the immediate project area and the entire Chuitna River watershed. Decreased groundwater recharge to area streams, lakes, and wetlands could impact fish and aquatic invertebrate populations in the entire region. These impacts and mitigation will be considered as part of the Fish Habitat Permit review process.

9. Human Uses: Game Management Unit (GMU) 16B is a state subsistence use area for Tier II moose hunt TM569 South Beluga including drainages south and west of the Beluga River, Beluga Lake, and Triumvirate Glacier. All of GMU 16B is also a federally recognized spring/summer subsistence migratory bird hunting and egg gathering area.

Local and non-local trappers have traplines in the area and aerial trapping and shooting of wolves also occurs in the area of the proposed mine. Ptarmigan and grouse are hunted by locals and non-locals in fall and winter throughout the project area. The Beluga area is a popular fall waterfowl hunting area for both local and non-local hunters. Access to these hunting and trapping areas is by vehicle along the local service roads, by fixed wing aircraft, and by snowmachine.

Personal use fishing occurs in local lakes and streams draining into the Beluga River. The Beluga River tributaries, Three-mile Creek and the Chuitna River are moderate to heavily used by recreational fishermen to catch rainbow trout, Dolly Varden Char, and primarily chinook and coho salmon. A large run of hooligan goes into the Beluga River. These fish are harvested on the beaches at Tyonek and Beluga.

Road, Conveyor, and Utility Corridor

10. Stream Crossings: The proposed road, conveyor, and utility corridor would cross three (3) streams that have been specified as being important to the spawning, rearing, or migration of anadromous fish. These are Threemile Creek (Stream No. 247-20-10002), an unnamed tributary to Threemile Creek, and Lone Creek (Stream No. 247-20-10010-2020). Fish Habitat Permits will be required for these crossings. All crossing structures, whether for road, conveyor, or utilities, should maintain the full hydrologic and ecologic functioning of the stream and maintain the integrity of the floodplain. Structures should span the full width of these streams and should utilize stream simulation design as a minimum standard. The portions of the conveyor that cross streams and wetlands should have containment structures to prevent the loss of coal, including coal dust.

11. Wildlife Movements: The connectivity of wildlife habitats for species such as moose and bears is very important. The wildlife baseline studies, as well as information from the ADF&G, should be used to identify important wildlife movement corridors so that these can be maintained. At a minimum, portions of the conveyor will need to be elevated or buried at identified travel corridors, and at stream and wetland crossings.

12. Wildlife-human Interactions: The development of the access roads and the increase in human activity that will accompany mine development will increase the potential for wildlife-vehicle collisions, allow access to presently remote areas, and may increase the harvest pressure on fish and wildlife resources in the project area. The SEIS should evaluate the impacts of increased human presence on wildlife populations in the project area. Mine and port operational plans should include measures to avoid the habituation of bears to humans or garbage. These plans, as well as protocols for dealing with habituated bears, should be developed in cooperation with the ADF&G.

Ladd Landing, Trestle, and Berth:

Development of the proposed mine will require a loading facility to transfer the coal to large, ocean going vessels for transport to secondary processing sites. We expect that the SEIS will

address potential alternatives to the proposed landing location, intertidal bulkhead, and approximate 2-mile long trestle.

13. Marine Mammals: The Cook Inlet beluga whale population is designated as a depleted stock under the Marine Mammal Protection Act (MMPA). The SEIS should address potential impacts of noise from construction and operation activities such as pile driving, the transfer of coal along the trestle, and increased vessel traffic in the area. The SEIS should also evaluate the potential impact of the bulkhead and trestle on local beluga movements and seasonal migration, and the potential impact of coal dust or residue that enters the marine environment and effects on belugas directly or indirectly through their prey.

The entire coastal area between the mouths of the Susitna, Ivan, Lewis, Theodore, and Beluga rivers south along the west shore of the inlet is heavily used by harbor seals for pupping and rearing of young, feeding on finfish and potentially some invertebrate species, and for local migrations during tidal inflow and outflow. Seals travel the waters along the shoreline on every tidal cycle throughout the ice-free months. There may be some opportunistic use of the area during winter months when ice conditions are light in order to feed on populations of tomcod that spawn in the lower reaches of the Chuitna River and possible Three Mile and Beluga River. Seal pupping and nursery haulouts are located at the mouths of the above mentioned rivers and are critical to pup survival.

14. Marine Fish: The proposed sheetpile bulkhead may significantly impact the movement and migration of juvenile and adult anadromous fish. Changes in current and loss of shallow water habitat could also lead to increased predation in the area of the proposed bulkhead. Potential alternatives to the proposed bulkhead that avoid and minimize impacts to marine fish such as a pile supported structure should be examined in the SEIS.

15. Terrestrial Wildlife: The shoreline beach is an important black and brown bear feeding area in the early spring months when runs of hooligan are beached during periods of moderate to heavy surf. Bears travel the beaches in search of stranded fish and other animals and feed on these sources of food. This is important during early spring prior to greenup of vegetation in snow-free areas. Throughout the summer months bears continue to use the beaches in search of food.

16. Human Uses: Nearshore, intertidal, and estuarine habitat such as that at the mouth of the Chuitna River are of great importance to larval and juvenile fish (both marine and anadromous) and shellfish. A broad range of invertebrate and groundfish resources have been, or currently are, harvested in both commercial and noncommercial fisheries in western Cook Inlet.

Commercial setnet fishing occurs along the beaches of the west side of Cook Inlet, including the area between Beluga and Tyonek, beginning in the last week of May and continuing through the summer months into September. Fishermen target Chinook during May and June, sockeye in June and July, and coho during August and September. Both local and non-local commercial fishermen fish in the area of the proposed trestle dock facility.

Subsistence salmon setnet fishing occurs on the beaches of the Tyonek Subdistrict between May 15th and October 16th. Residents of Tyonek, Beluga, and Shirelyville obtain permits for setnet fishing on designated days throughout the season.

Tyonek residents who are Alaska Natives qualify under the MMPA to harvest harbor seals and beluga whale. Tyonek residents take harbor seals as part of their annually subsistence harvests. The village of Tyonek has an allocation (1 or 2 whales) under the new beluga harvest restrictions on Cook Inlet beluga whale.

The SEIS should evaluate the potential impacts to harvest activities that may occur as a result of: (a) upstream and down-stream sedimentation resulting from a solid fill intertidal dock facility; (b) changes to fish and marine mammal migration patterns; (c) boat restrictions associated with loading activities near the trestle and vessel terminal that may limit and/or restrict small boat access to beach sites and inlet waters; and (d) increased wave action generated by large cargo vessels.

17. Coal Dust: The SEIS should address the potential impacts of coal dust to water quality in Cook Inlet from the proposed stockpiles at the Ladd landing facility and from the portion of the conveyor that extends into the marine environment. Alternatives such as covered stockpiles and a fully enclosed conveyor should be examined.

That concludes our scoping comments for the development of the Chuitna Coal Project SEIS. Please contact me at (907) 269-8629 with any questions.

Sincerely yours,

Rick Fredericks for

Tom Crafford
Large Mine Project Coordinator
(907) 269-8629, phone
(907) 269-8930, fax

cc: Skip Joy
Phil Brna