Reconnaissance Phase

Tar Creek and Lower Spring River Watershed Management Plan

Mine Drainage Control and Treatment

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Mine Drainage Control and Treatment

Reversing the impacts on surface and ground waters (the water environment) of lead-zinc mining within the Tar Creek – Lower Spring River Watershed involves understanding and countering the causes by considering:

- Water impacts created by the various phases of the mining life-cycle and their remedial significance,
- The drainage and water mass balance changes imposed by mining and corresponding hydrologic consequences,
- How the hydrologic system is currently working, and
- Desired post-mine land use and watershed function goals contrary to the particular detail mine problem areas within the Oklahoma portion of the Tri-State Mining District.

This section describes the general nature of the mining wastes and there contribution to water problems, mining impacts on the water environment from the various phases of the mining life-cycle with reference to the Tar Creek – Lower Spring River Watersheds, and possible remedies to these mine drainage impacts in the Tar Creek area.

In addition to lead-zinc metal mining impacts, the upper portions of Spring River in Kansas and Missouri have been impacted by coal mining, much of which received little reclamation attention having occurred before stricter Coal Mining regulations were imposed in the late 1970s (Surface Mining Control and Reclamation Act, SMCRA, 1977; P.L. 95-87).

Existing and complementary projects will address the various aspects of these remedies in a coordinated and integrated manner to minimize rework, maximize improving the water environment's function, and to allow application of adaptive management strategies to forward watershed enhancement, public health and safety, and future economic viability within the watershed.

The following discussion is provided for those unfamiliar with the impacts of mining on the water environment which is followed by specific information regarding hydrologic impacts and considered remedies (features and options) within the Tar Creek – Lower Spring River Watershed.

Mine Wastes and Tailings

Various terminologies are in use regarding the chat, milling waste and other mine spoil found in the Oklahoma portion of the Tri-State Mining area. For term of reference, mine wastes can generally be classified into two major categories based on the mining and mineral processing cycle (Younger *et al*, 2002; see Figures 1 and 2):

- Waste rock or Spoil, and
- Tailings (also know as "finings" in the coal sector).

As shown in Figure 1, waste rock is generated both before and during mineral extraction, and during mineral processing. By contrast, tailings arise solely in the course of mineral processing. Wills (1992) presents a comprehensive review of mineral processing techniques. Because of the availability of this review and because it is not necessary to understand all the details of mineral processing to appreciate the environmental significance of tailings, suffice it be said that the process of converting run-of-mine ore into a marketable commodity generally involves the following sequence of activities:

1) Primary screening (i.e., large-scale sieving) of the run-of-mine rock to remove large blocks of country rock,
2) Crushing of the ore to sufficient fine grain size that virtually all grains of ore minerals will be free of gangue (nonmetal or unvaluable minerals in the ore).

3)

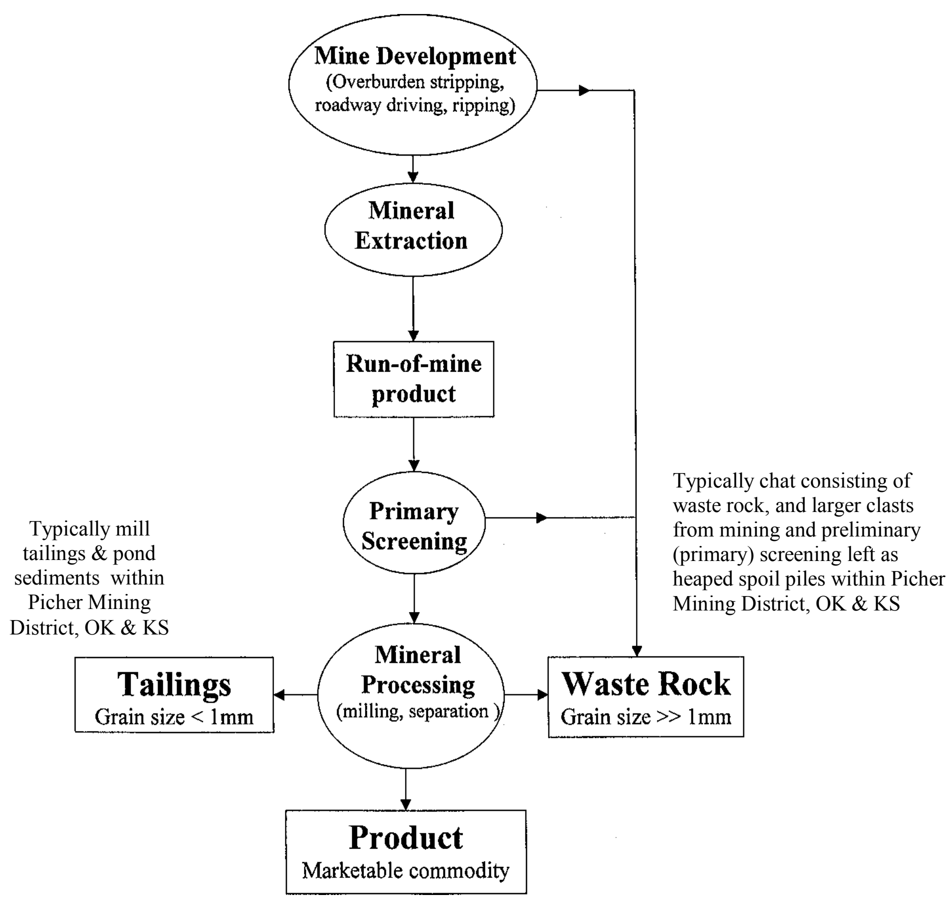

Figure 1. Mine waste classification according to source generation from the mining and mineral processing cycle. [Adapted from Younger et al, 2002, Fig. 1.28)

Separation of the ore and gangue minerals.

Waste rock and tailings both result during this sequence of activities, with earlier phases generating most of the waste rock, and the final step releasing tailings (Fig. 1). Genetic considerations aside, the fundamental distinction between waste rock and tailings is one of grain size, with most of the tailings being finer than 1-mm in diameter, and most waste rock being considerably coarser than this (Figure 2).

The difference between waste rock and tailings are summarized as follows:

Waste Rock Characteristics	**Tailings Characteristics**
Predominately coarse-grained (1-50 mm)	Predominately fine-grained (<1 mm)
Moderately reactive if sulfidic	Highly reactive if sulfidic
Moderate to high permeability en masse	Low permeability en masse
Generally placed dry	Generally deposited from flowing water

The contrast in grain-size between different waste rock materials, and between waste rock and tailings are shown in Figure 2. Processing by Eagle Picher Mining & Smelting Company at the Central Mill actually provided “custom milling.”

The Central Mill was the largest and most efficient lead-zinc concentrator in the Tri-State District. Several million dollars was spent on design and equipment to ensure that custom ores were accurately weighed, sampled and accounted for, and that maximum recovery of minerals were obtained. For this reason, the Shippers to the Central Mill, and the Fee

Owners of the land from which the ore was mined realized a larger remuneration than they would if the same ore were treated in the conventional Picher-type jig mill (Eagle Picher, 1951). Comparison data on these two milling processes are shown in Table 1.

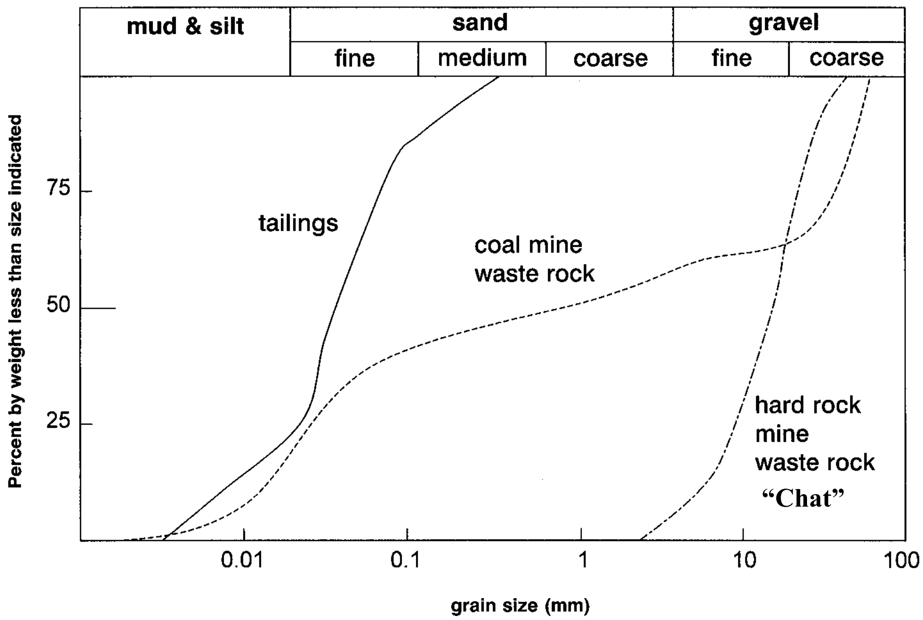

Figure 2. Median grain-size distribution curves for typical tailings, coal mine waste rock and typical 'hard rock' metal mining setting. Lead-zinc mining is in the hard rock category of waste rock production. [Adapted from Younger, et al (2002) with plots used from Poulin, et al, 1996 and NCB (1973).

Table 1
Central Mill vs Picher-type Jig Mill Comparative Data
(Eagle Picher, 1951, pp 2-3)

Transportation of mined ore by rail service furnished to established mines without charge. Truck service furnished to others at actual cost.	Only truck service available.
Facilities for the accurate determination of the metal content of the mined ore are provided. These include accurate automatic weighing, accurate moisture sampling, a specially designed sampling and sample preparation plant and a modern assaying laboratory operated by trained technicians.	No head sampling provided.
Automatic sampling and accurate analyses of all mill residues are made, including cone tailing, jig tailing, flotation tailing and thickener overflow; also accurate sampling and analyses of the zinc and lead concentrates are made from which a complete metallurgical accounting of the minerals in the mined ore is prepared.	No such facilities provided.
In the Central Mill the latest metallurgical developments for the treatment of Tri-State mine ores are used, such as:	
Heavy-Media Separation (Sink-Float). The most recent improvement in separation of ore values in coarse sizes.	Old Harz type jigs inefficient by present standards.
Bendelari jigs--most efficient method in separation of ore values in fine sizes.	Old Harz type jigs inefficient by present standards.
Careful desliming of jig feed and tailings.	Inadequate desliming.
Large and adequate thickener capacity.	Small and inadequate thickeners.
Fine grinding through use of large ball mills.	Grinding by rolls or inadequate grinding in small ball mills.
Modern efficient flotation machines.	Old type flotation machines.
Filtering of lead flotation concentrate to eliminate loss of fine lead.	Tabling of lead flotation concentrate with consequent loss of lead in slime.
Metallurgical supervision by experienced graduate metallurgists with special attention to maintenance of highest grade of zinc and lead concentrate consistent with maximum commercial recovery of mineral values.	Often no metallurgical supervision.
A detailed statement is submitted each month to the shipper and landowner covering their mine ore received at the Central Mill. The average assay of each shipment, with monthly metallurgical results, is used to calculate the value of the ore by a formula developed by the U. S. Bureau of Mines; whereby all concentrate production from the Central Mill is accounted for, and each shipper receives the correct value for his mined ore.	It is impossible to have complete segregation of ores from different shippers in the present Picher-type mill. When ores of varying values are being milled the richer ores are penalized to the benefit of the leaner ores because of variations in percentage of values picked up or deposited in the mill circuit at such points as bedding on jigs, elevator boots and conveyors. This loss or gain in shipment values cannot be prevented without accurate head sampling, accurate sampling of all mill residues and concentrates and complete metallurgical balances. Consequently settlements for shipments are made only on estimated values without adequate information.

These observations lead to several conclusions regarding the hydrologic impacts of these processes:

- Source rock lithology is at least as important as processing technology in determining the grain-size distribution of waste rock and tailings.
- The specific outcome of processing a particular ore is difficult to predict accurately.
- The thoroughness of processing is directly proportional to the residual mineral material available for future leaching or other chemical transformation or physical transport.
- The smaller the size of such material, the greater the secondary mineral leaching potential.

Consequently, it is recommended that site-specific grain-size measurements be made whenever mine wastes are the subject of crucial management decisions.

Disposal of Waste Rock and Tailings

Where waste rock cannot be back-filled into the mine void, or else exported from the mine site for use as bulk fill in construction projects, it will generally be disposed of in a waste rock pile (or "spoil heap"). Waste rock piles are generally formed by loose tipping (or "end tipping"), from trucks or conveyor belt systems. The process of end-tipping results in the development of significant variability in the sediment fabric (Figure 3). Gravity sorts end-tipped wastes as they roll down the face of the heap (which typically conform to angles of repose of around 40°). Large blocks and cobbles roll to the foot of the slope while finer sediment remains near the top of the slope (Figure 3a). This phenomenon has been widely observed (e.g. Winczewski, 1979; Groenewold and Rehm, 1982; Hawkins, 1994; Newman et al., 1997). Repeated end-tipping results in the accumulation of inclined layers of systematically graded rock fragments. If the flank of a waste rock pile is excavated, it often reveals an internal structure akin to that shown illustratively in Figure 3b (see also Winczewski, 1979; Newman et al., 1997, and Wilson et al., 2000). This has significant hydrological implications.

Most waste rock is emplaced without deliberate compaction. In deep mine spoil heaps, which are typically formed by end-tipping from conveyor belts, compaction under the weight of the spoil itself may be the only operative compaction process.

Revegetation of waste rock piles is now common practice in the mining industry world-wide. Considerable research has been undertaken into the best way to stimulate plant growth and pedogenic (soil-forming) processes on reclaimed spoil (e.g. Rimmer and Younger, 1997). However, until recently far less attention was paid to ensuring that the drainage of reclaimed waste rock piles was arranged such that it minimized leachate (acid mine drainage or AMD) generation in the long-term. Consequently, many visually attractive spoil heaps which were reclaimed in the 1970s are now giving rise to polluting discharges of acidic, metal-rich leachate (see for instance Younger et al., 1997, for unreclaimed spoil piles, see water quality data for 1980 to 1989 from Tar Creek Task Force Report, 2000).

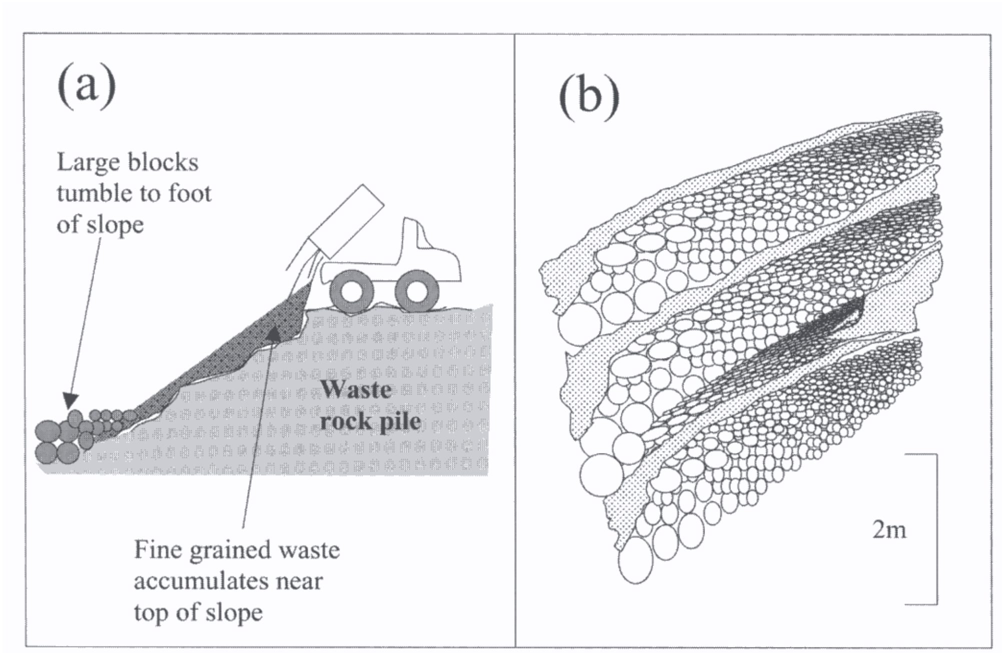

Figure 3. End-tipping as the source of heterogeneity in mine waste rock. (a) End-tipping, resulting in sorting of the large and small fragments as they roll down the tip face. (b) Graphical representation showing the fabric (texture) of the waste rock accumulated from end-tipping by truck, conveyor or other means. [Adapted from

Until the early 20th Century, virtually all tailings were simply released into the nearest watercourse (e.g., see Pirrie et al., 1997). Where mining continued for many centuries, this practice sometimes led to watercourses and estuaries becoming clogged with tailings and when used in connection with transporting the ore, they became no longer navigable. In addition, complaints from riparian downstream owners from mineral processing plants led to the development of alternative tailings disposal methods. With the emergence of greater environmental awareness in the last fifty years, these methods became better developed.

By the 1940s, the disposal of tailings in sedimentation ponds or lagoons specifically for that purpose was common practice in the mining industry worldwide (Younger et al, 2002). Sedimentation ponds are known by various names, such as “tailings dam” or “tailings dike (dyke in Europe, Vick, 1983) and “slimes dam” (South Africa, e.g., see Bright, 1997). Despite this embrace of tailings dam technology, there still are small or medium size enterprises (SME) and informal sector mining operations in remote areas continuing the practice of stream-disposing tailings.

Tailings dam technology is unaparently simple. There are two main types of tailing dams – ring dikes and valley dams – and three methods of dam construction (Vick, 1983; Younger *et al*, 2002). Ring dikes are constructed in flat lying areas, and consist of a multi-sided dike entirely around and enclosing the sedimentation pond (or lagoon).

Valley dams are tailings dams which are constructed by impounding a natural valley or ravine, or part of a valley or ravine. Details of the design of tailings dams are provided by Vick (1983), while Klohn (1979) provides a comprehensive discussion of design procedures and operational interventions to minimize problems of seepage through dikes.

While the majority of tailings dams are constructed and operated in accordance with the above principles, the vast majority of which function according to design and prevent environmental pollution, there are those (some spectacularly) that fail. Several worldwide surveys of tailing dam failures have concurred that the root cause of the increased rate of failure is not poor design, but rather stems from inadequate implementation of the original design (UNEP, 1996; ICOLD, 2001, Younger et al 2002). This occurs because unlike other dams, tailings dams are constructed gradually over many years or even decades, so that the site operators in the later phases of construction are unlikely to be the same personnel as those who originally designed the dam many years previously. How best the scope for human error can be “designed out” of tailings dam construction is still a matter of active debate within the mining industry (Younger, et al, 2002).

From a hydrologic impact perspective, tailings dams failures have a common causative agent which is water (UNEP, 1996; ICOLD, 2001). The two most common causes that water promotes dam failure are:

- Through erosive down-cutting of dikes if flows exceed spillway capacities (Smith and Connell, 1979; ICOLD, 2001), or
- By promoting slippage due to liquefaction or sapping of the dike core by seepage waters (Robinson and Toland, 1979; Smith and Connell, 1979; UNEP, 1996).

Wet conditions downstream of a failed dike also greatly increase the movement of destructive sediment or mudflows of tailings (Blight, 1997). These in turn can choke steam channels with sediment which subsequently make there way downstream through later periodic flooding events and can divert flows or lead to ponding within channel reaches changing the flow regime with that reach of stream.

As with waste rock piles, drainage of leachate (AMD) from both active and abandoned tailings dams can be a significant source of surface water pollution, as well as a potential source of structural instability over the long term.

Luza (1986) compiled the disposition of mine and mill wastes as of 1982 occurring within the Tar Creek Watershed area in Oklahoma. Approximately 2,900 acres in Oklahoma were overlain by mine and/or mill by-products. The discarded mill-waste (tailings) are primarily composed of chert fragments with a grain size less than or equal to 0.75 inches (19.05 mm) and referred to as “chat.” Chat was transferred from mill sites by a series of conveyors and elevators and heaped into piles. Pile heights vary from 28 to 205 ft high. Prior to 1930, most mines had their own mill, which prompted disposal on every greater areas of agricultural land. Ore recovery prior to 1920 varied from 58-70%. Tailing piles were reworked later with the advance of improved extraction metallurgy at least twice and even a third time for their lead and zinc content.

From the 1982 inventory, there were 146 former chat-pile sites and 119 existing chat piles covering approximately 1,200 acres. There were in addition, 14 tailing ponds. About 900 acres are overlain by chat piles and there were 33 chat piles containing 95% available chat for commercial use. Some of the larger chat piles were being reworked (as of May 1980, Luza reported about 8 chat pile areas were being worked either actively or intermittently, Luza, 1986, p. 16 and Table 7, p. 19). Common uses of the material include railroad ballast, road foundation material, and concrete and asphalt aggregate.

Approximate volume of the 33 chat piles (as of May 1980) was 45.61 million cubic yards (yd^3) with volumes of individual piles varying from 80,000 to 6.9 million yd^3 (Luza, 1986, p. 17). This is equivalent to 61.5 million short tons (assuming a bulk density of 100 lbs/ft^3). There were 87 minor tailing (mill-waste) piles containing 30,000 cubic yards or less of chat per pile. Minor chat piles were not included in this total (Luza, 1986, p. 17).

Tailing ponds cover about 800 acres in the Oklahoma portion of the Picher Field. Almost every mill site had a tailings pond but only the major tailing ponds were inventoried by Luza. The Eagle Picher Central Mill had two of the largest ponds covering 79 and 148 acres, respectively. Approximately 260 acres (22%) of the total acreage occupied by chat piles had been reclaimed. The reclaimed land use is supportive of varies forms of agricultural activity. Some sites are reported to have urban land uses: residential housing and light industry activity (salvage yard) (Luza, 1986, p. 20).

Mining Impacts on the Water Environment

Mining impacts on the water environment arise from five distinct phases of the mining life-cycle. Six types of negative impacts on the water environment occur:

1) From the mining process itself,
2) From mineral processing operations,
3) From the dewatering activities which is undertaken to make mining possible,
4) From seepage of contaminated leachate (acid mine drainage or AMD) from waste rock piles and tailings dams and runoff,
5) During flooding of workings after extraction has ceased,
6) By discharge of untreated waters (AMD) after flooding is complete.

All these impacts have been experienced within Tar Creek and the Lower Spring River Watersheds. Of these six sources of impacts, the first is usually the least important, and the last is certainly the most important in the long term. For Tar Creek prior to mine closings, sources (2) and (3) were serious concerns. Since mine closings in the early 1970's, the last three have been of greatest concern. Actually the last three types of impacts are still occurring, albeit at varying rates related to water availability and wet-dry cycles occurring since mining ceased and natural hydrologic processes adapt to the man-made terrain and subsurface geologic alterations.

Seepage of contaminated leachate (AMD) from waste rock piles and tailings dams is a significant cause of surface water pollution in many mining districts. Numerous instances are known where waste rock piles which were thoroughly re-vegetated several decades ago continue to release acidic leachates or acid mine drainage (AMD), emanating from shallow water table systems perched within the spoil (e.g. Younger et al., 1997).

Drainage of leachate through the unlined bases of old tailings dams is also known to give rise to pollution of both surface waters and ground waters (Manzano et al., 1999; Johnson et al., 2000). Besides seepage releases, unreclaimed spoil can give rise to highly polluted surface runoff during storms (e.g. Bayless and Olyphant, 1993). This was experienced during the 1987 to 1989 period within the Tar Creek Watershed as will discussed later below.

Mine Flooding and Groundwater Rebound

The flooding of abandoned mine workings after mining has ended has long been called "*water table rebound*" (e.g. Henton, 1979, 1981), or "*groundwater rebound*" (e.g. Robins, 1990; Smith and ColIs, 1996; Sherwood and Younger, 1997). The process of rebound in deep mines commonly results in a marked deterioration in the quality of mine waters (e.g. Cairney and Frost, 1975; Younger, 1993a, 1998b,c, 2000a,b). The reason for this is the sudden dissolution of so-called "acid-generating salts" (AGS), which is the name given to a collection of secondary minerals formed by the weathering of pyrite above the water table (Bayless and Olyphant, 1993; Younger, 2000a,b). The AGS accumulate within the dewatered workings wherever oxidizing pyrite is out of reach of flowing water. As the workings flood, all the AGS between the deepest level of mining and the rest water table position will eventually be dissolved as the water surface reaches them. Younger (2000a) has noted that this commonly results in a ten-fold increase in the concentrations of contaminants (particularly iron) within the mine waters (Younger et al, 2002).

Similar deteriorations in quality are sometimes observed following the initial saturation of some strip mine back-fill materials after restoration (e.g. Marsden *et al*., 1997; Younger, 2000c), and during the flooding of open-pit mines to form pit lakes (Geller et al., 1998). This has occurred in the upper portions of the Spring River Watershed in Kansas and Missouri due to past coal mining activities prior to 1977 federal regulations (Marcher and Kenny et al, 1984).

Such deterioration in subsurface water quality has important environmental consequences in the subsequent phase in the mine life-cycle, when the abandoned workings overspill to the surface environment. This is actively occurring at Mayer Spring and the Tar Creek-Lytle Creek confluence as well as near Beaver Creek in Spring River Watershed.

Besides these chemical changes, the rebound process can also prompt physical changes within the mined system. At least four scenarios have been documented:

1) Subsidence as open voids are eroded, either by rapidly flowing water winnowing support pillars, or perhaps more commonly by pillar collapse as the wetting of seat-earths (i.e. weak mudstones which often underlie coal seams) leads to slaking and a loss of strength. For instance, surface subsidence features which appeared in the recently-abandoned Leicestershire Coalfield (central England) in the early 1990s have been ascribed to the collapse of an 150-year old barrier pillar when it

was inundated by rising mine waters (Smith and ColIs, 1996). This also can occur in karst areas where limestone and dolomite are eroded by dissolution to form sinkholes or collapse features (see ***Subsidence and Collapse Features*** discussion below).

2) Fault re-activation has been recorded during the flooding of deep mine workings in a number of European countries (Donnelly, 2000). The increase in pore pressures which occurs during rebound can reduce the frictional resistance to movement in extensional fault planes, leading to a temporary increase in seismicity, sometimes with associated property damage.

3) Rising ground levels of up to 25 cm have actually been recorded above an area of actively-flooding workings in the Limburg coalfield of the southern Netherlands (Bekendam and Pottgens, 1995). While the exact mechanism is not known, it is suspected to involve the re-hydration of swelling clays in the sequence, and an increase in buoyant support of open voids.

4) Development of high pressure gas pockets has been directly observed by miners who have been temporarily trapped in high points within deep mine workings, many meters below the water table, during catastrophic flooding incidents (e.g. Llewellyn, 1992; Younger and LaPierre, 2000). In one recent case (Younger, 1999; Younger and LaPierre, 2000), audible explosions originating at many hundreds of metres' depth in actively-flooding tin mine workings were ascribed to pneumatic fracturing of the crowns of open stopes as the pressure of gas trapped by rising mine waters exceeded the strength of the rock mass. The perceived sequence of events is illustrated diagrammatically in Figure 4.

The violent venting of air or other mine gases trapped in high pressure pockets during rebound is only one aspect of a more general process of acceleration of mine gas emissions during rebound. Low density mine gases such as methane will tend to behave like air: they will either vent readily from the workings if the pathways are available, or else they will accumulate under increasing pressure in high spots in the workings. High density mine gases such as hydrogen sulfide, which was encountered in much of the Picher Field when mine dewatering was first started, by contrast, will tend to form an invisible blanket "floating" on the surface of the rising mine water. This is the expected behavior of some important hazardous mine gases, such as radon and stythe (CO_2 - rich gas). The rising water table therefore tends to drive mine gases before it. This is a point worth remembering for anyone contemplating exploring old workings which do not receive forced ventilation. A tragic case which occurred in Northumberland, UK, in 1995 illustrates the dangers. In this instance a man working in a small factory was overcome by carbon dioxide when he entered a well-frequented basement area, which happened to adjoin an old drift portal (Burrell and Friel, 1996). The hitherto unprecedented carbon dioxide accumulation in this area has been ascribed to a local rise in the water table in the old workings (Younger *et a*l, 2002).

One indirect impact of the rebound process is that, as long as it continues, previously-pumped waters will no longer be released to the surface environment. Where these pumped waters had been useful (for instance in providing dilution for sewage effluents

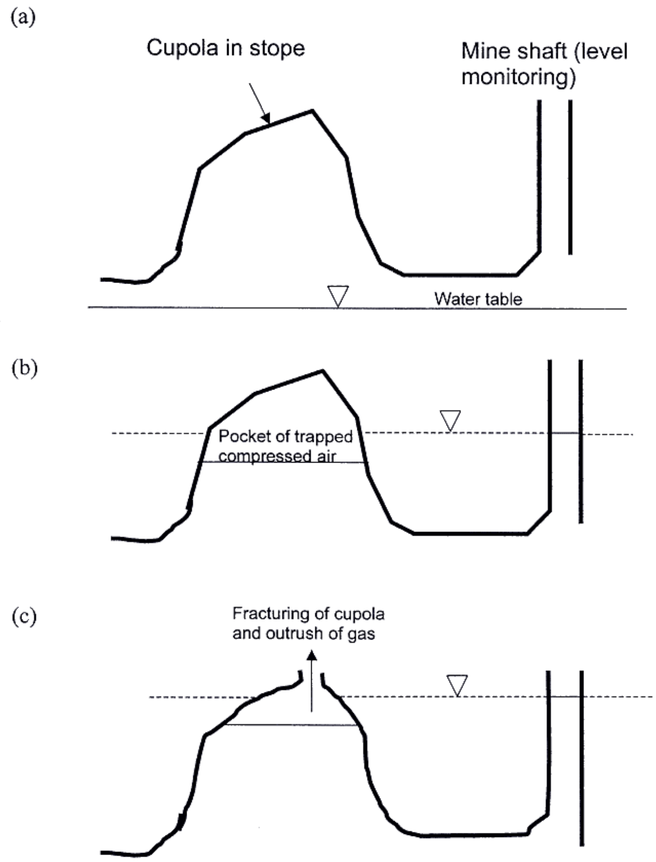

Figure 4. Possible process of pneumatic fracturing of a cupola in a large open stope, as is postulated to have occurred during mine water rebound in South Crofty Tin Mine, Cornwall (UK), in late July 1998 (after Younger and LaPierre, 2000). [Adapted from Younger et al, 2002).

entering the same rivers), then this temporary loss of surface flows can be significant (Banks et al, 1996).

The *discharge of untreated mine waters after the flooding of workings* can lead to:

- Surface water pollution
- Pollution of over-lying aquifers
- Localized flooding, and
- Over-loading and clogging of sewers

While the first impact may be obvious, the other three impacts are not as well documented:

- ✓ Pollution of over-lying aquifers by upward migration of mine water has been recorded far less frequently than might be imagined. Only one unequivocal case has been published, in which part of a public-supply dolomitic aquifer in northeast England was polluted when underlying coal mines were allowed to flood in the late 1970s (Younger and Adams, 1999).
- ✓ Localized flooding has affected residential, agricultural and industrial land in at least four UK coalfields (Younger and Adams, 1999), wherever new mine water discharges have emerged onto previously dry land. In many cases the flooding is a side affect of the following process:
- ✓ Over-loading of sewers occurs when the volume of water routed into a sewer exceeds the design flow. In some cases, high-volume mine water discharges have so overwhelmed the capacity of sewers that surface flooding has resulted (e.g. Younger, 2000c). Even if the mine water flows are not excessive, the clogging of the sewers with ferric hydroxide (ochre) precipitates can reduce the effective diameter of the sewer sufficiently that water backs-up and flooding occurs. This has been observed in Scotland (Younger, 2000c) and in Cleveland, UK (Younger, 2000d).

Subsidence and Collapse Features

Luza (1986) has compiled the most detailed evaluation of stability problems associated with the underground mine workings of the Oklahoma portion of the Picher field. Approximately 2,450 acres within and adjacent to the Picher-Cardin, OK area are underlain by abandoned lead-zinc underground mines. At least 1,064 shafts were in the Oklahoma portion of the Picher Field with 481 (45%) of the mineshafts either open or in various stages of collapse. More than 50% of these mineshafts were reported by Luza (1986) as being either concealed or filled. Of the sites having access, 316 or about 65% of the sites could be filled. These sites include open shafts with minor collapses (≤ 10-ft diameter) as well as non-shaft areas having minor (2-30 ft diameter) to moderate (31-94 ft diameter) collapses. Luza's inventory summary for shafts and surface collapses is provided in Table 2.

Open shafts and surface collapse features associated with the underground mine working represent one of the greatest visible hazard potential in the Oklahoma portion of he Picher Field. The initial field inspection program was conducted from May 1981 to May 1982.

The 1986 findings are currently being updated with changes being verified by Luza and Keheley (Luza, 2004, personal communications).

Collapse history of the mine site was also compiled in the 1981-82 field inspection and correlated through time by the use of aerial photography taken in 1939, 1952, 1964, 1972, 1979, 1980 and 1982. These results are shown in Table 3. All major collapses (both shaft and non-shaft related) were studied by Luza in detail. Twenty-nine major collapses associated with 34 shafts and 55 non-shaft related collapses with most (66%) of the collapses occurring prior to 1952 when the mines were still active. There was only one new collapse site found to have occurred after 1979 (20-ft diameter collapse SE of Commerce, OK), and several existing collapses were enlarged since 1979.

Table 2
1986 Shaft And Collapse Status Inventory
(after Luza, 1986, Table 5, p. 13)

Shaft Status	Diameter (ft)	Number	Percent of Total
Open		59	6
Open with minor collapse	≤ 10	36	3
Minor collapse	2-30	241	23
Moderate collapse	31-94	115	11
Major collapse	>95	30*	3
Concealed filled		558	52
Covered		25	2
Totals		**1,064**	**100**

* At three sites, two shafts were involved in the same collapse; collapse is listed twice for each site. At one site, three shafts were involved in the same collapse; collapse is listed three times. Hence, 30 shafts are involved in 25 separate collapses.

Approximately 27 surface acres were disturbed due to the shaft-related collapses. Major shaft-related collapses account for more than 80% (22 surface acres) of the total disturbed land (Luza, 1986). Four shaft-related collapses covering 3.3 acres near Commerce, OK were filled between 1964 and 1979. From the last two columns of Table 3, collapse events actually increased between 1952 to 1979 when during the later portion of this time span (after 1970) mine workings began to recharge since major mine dewatering activities ceased. Mines rebounded by the early 1980s. This likely explains why major collapse events have slowed since water supports large areas of the former underground workings.

Table 3
Time Related Collapse History, 1939-1979
(Adapted from Luza, 1986, Table 6, p. 15)

Time Interval	Number of Collapses * (Approx. years in interval)	Interval Percentage	Collapses per yr	Yrs between Collapses
(1902) Pre-1939	27 (37)	33	0.73	1.37
1939-52	25 (14)	30	1.79	0.56
1953-64	9 (12)	11	0.75	1.33
1965-72	8 (8)	10	1.00	1.00
1973-79	14 (7)	17	2.00	0.50
Totals	**83**	**100**	**Ave = 1.25 ± 0.6**	**Ave = 0.95 ± 0.4**

* Collapse features described by Luza (1986, Table 3)

The flooded underground mine workings and associated collapse/subsidence phenomena within the Picher Field are not unlike that of an artificial or man-made karst terrain. Karst terrain results from the development of solution cavities within carbonate formations (limestone /dolomite, gypsum or other evaporite) where in this case the cavity development was assisted by mine workings within certain limestone/chert/dolomitized host rock of the Mississippian-age Boone Formation. In this regard development of subsidence or collapse features can be explained from a karst-like process.

The terms subsidence and collapse usually are used as synonyms. The terms sinkhole and doline refer only to localized land surface depressions arising from karst processes (Back, 1984; Milanović, 2004).

Subsidences are spatially independent, random occurrences and have been identified as sources of major potential problems, which may cause considerable damage in reservoir bottoms, in urban areas, at industrial sites, and near communication lines. The catastrophic nature of subsidence development is unpredictable and practically instantaneous and therefore very harmful.

Subsidence is common when karstified rocks are covered with unconsolidated sediments. This occurs under the influence of water, as the erosion and piping action breaks down the support of poorly consolidated sediments. The destructive role of water can be distinguished in four different categories:

- Underground water acting from below
- Surface water acting as floodwater
- Pore water within alluvial deposits (overburden)
- Water acting indirectly, pressurizing the air in the aeration zone

The genesis of subsidences is usually related to these categories of processes.

Milanović (2004, p. 25-27) describes a generalized and simplified model of these processes (refer to Figure 5).

"Initially, the conduit opening (ponor) was formed by karst processes acting within carbonate rocks or in other rocks susceptible to karstification. During the process of sedimentation, the opening was covered with deposits of clay, terra-rosa, sand, gravel, cobbles, and boulders. Under the influence of moving water, the process of mechanical suffusion[1] destroyed homogeneity of unconsolidated deposits. As a result of this destruction, a relaxation zone formed, shaped approximately as an arch, semi-ellipse, or parabola (**I**). This surface will remain unchanged as long as external forces are in equilibrium with resisting arch forces. Internal friction, specifically shearing resistance along vertical planes, plays an important role in maintaining this equilibrium profile. However, surface waters, groundwaters, or pore waters essentially change the cohesion and angle of internal friction of the soil [overburden], and a new load release surface is formed (**II**) according to the new state of equilibrium. Part of the material between spherical surfaces (arches) (**I** and **II**) will subside into caves and be transported by water further away in the karst system. When the roof of the spherical surface approaches ground

[1] Suffusion is the process of spreading a fluid into the surrounding media.

surface, the entire roof caves in, thus producing a circular opening on the land surface (alluvial ponor or in the case of shale or limestone a collapse feature)."

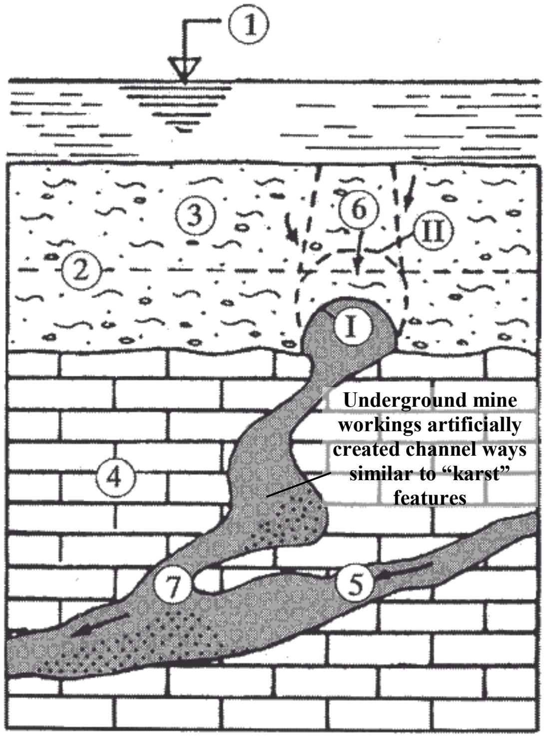

Figure 5. Formation scheme for subsidence: (1) Flood level; (2) groundwater level; (3) overburden: unconsolidated alluvial sand or other clastic deposits (shale and sandstone overlying Boone Formation); (4) carbonate rocks; (5) direction of groundwater flow (within mine workings); (6) filtration direction of floodwater or infiltrating surface water source (pond, etc.); (7) karst channel or artificial channel developed by underground mining in carbonate terrain. [Modified from Milanović (2004, p. 26-27]

The origins of subsidences can be either natural or induced. Induced subsidence is a consequence of the urbanization, mining, reservoir construction, and groundwater extraction that dramatically increased in the second part of the twentieth century. Luza (1986) reported several collapses in the late 1960s and 1970s effecting people, structures and a road in Picher. Failure surfaces dropped between 25 to 50 feet.

According to LaMoreaux and Newton (1986), thousands of collapses have formed in the U.S. since 1950 with more than 80% of the identified events a consequence of uncontrolled pumping of ground water for water supply or other purposes. According to the United States Geological Survey (USGS), in 1991 the national Research Council estimated that annual costs in the U.S. from flooding and structural damage caused by land subsidence exceeded $125 million (USGS, 1991). Indirect costs are many times higher.

Mine Water Chemistry

The minerals and coal that represent economically valued ores are largely chemically stable under natural, in-situ geological conditions. When excavated by mining and exposed to the atmosphere, however, these solid phases become chemically unstable. In the case of coal and hydrocarbons, they can be oxidized through combustion to gain useful energy. The sulfide minerals associated with metal ores and present as sulfur contamination in coal (or carbonaceous material) will spontaneously dissolve when in contact with water. The chemical weathering of sulfide minerals represents a series of linked geochemical and microbiologically-mediated reactions through which contaminants are released from ore and associated wet-dry zones within underground or surface workings, mine waste rock and mill tailings into the hydrological cycle becoming mobile and thus bioavailable as potentially toxic solutes.

Table 4 shows the primary and secondary minerals for the Tri-State Mining District which contribute to the geochemistry of these processes.. Release of metal ions to acidic waters results from the weathering of sulfide ores containing minerals such as sphalerite ($ZnS(s)$), galena ($PbS(s)$), wurtzite ($ZnS(s)$) and chalcopyrite ($CuFeS_2(s)$). *Acidity* is primarily released during the weathering of pyrite ($FeS_2(s)$ in solutions containing dissolved oxygen (O_2). Subsequent *precipitation* of iron under oxygen-rich (oxic) conditions produces iron oxyhydroxides minerals. These precipitates are generically termed *ochre*[2] which is visibly deposited as yellow to red-brown coatings on solid surfaces within effluent channels at mined sites. Tar Creek and Lytle Creek as well as their unnamed tributaries in the mining area exhibit this in most of their reaches.

Metal ions, acidity and ochre all represent significant environmental hazards to freshwater resources. Additional Activities' remedial actions and environmental risk assessment of abandoned sites must consider:

- the strength of the contaminant source (contaminant flux),
- the intensity of contamination (contaminant concentrations),
- the duration of the source (contaminant lifetime) and
- natural attenuation processes.

In addition, hydrological transport pathways to sensitive receptors such as valued surface waters and groundwater abstraction zones must be considered. Figure 6 illustrates the various features of the mine workings, corresponding contaminant sources, possible receptors, transport pathways and possible attenuation processes.

The following chemical reactions represent processes that produce acidity, ochre and soluble metal ions when sulfide minerals dissolve in the presence of dissolved oxygen. These weathering processes are examples of redox (*or electron transfer*) reactions.

Transfer of electrons to dissolved oxygen (O_2) from sulfur (S) and iron (Fe) in the mineral phases, transforms S and Fe into soluble forms. The flow of electrons that occurs

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[2] The term "ochre" is commonly used as a collective term for the red, orange and yellow iron salts produced by hydrolysis of ferric iron. The precise composition of ochre varies with pH and the availability of dissolved anions such as sulfate ($S0_4^{2-}$). Under near neutral conditions (pH ~ 6-8), a mixture of X-ray amorphous iron hydroxide and goethite (α-FeOOH) precipitates. At more elevated pH (> 8) ferrihydrite ($Fe(OH)_3$) is more commonly precipitated. At lower pH conditions, substitution of sulfate ($S0_4^{2-}$) for hydroxyl ion (OH-) occurs, resulting in the formation of "oxyhydroxysulfate" minerals such as schwartmanite (Winland et at., 1991; Bigham et at., 1992). If left to consolidate over many years (or perhaps decades) the various hydrated ferric iron salts which comprise ochre will gradually dehydrate and recrystallize to form hematite (Fe2O3)' which is a major iron ore mineral. Haematite is far denser than the hydrated ferric iron salts, a fact which has potentially important implications for the long-term performance of passive systems in which ochre is presently accumulating [Younger at al, 2004].

Table 4
Minerals of the Tri-State Lead-Zinc Producing Districts (KS, MO, OK)
(Bastin, E. S. (1939), paper by Bastin, E. S. and Behre, Jr., C. H., Tables 3 & 4; and McKnight & Fischer (1970).

PRIMARY MINERALS

Elements
- Silver (native?)

Sulfides
- Sphalerite ✓(1) [ZnS]
- Wurtzite (primary?) ✓(2) [ZnS]
- Galena ✓(3) [PbS]
- Chalcopyrite ✓(4) [Cu Fe S_2]
- Marcasite ✓(7) [FeS_2]
- Pyrite ✓(8) [FeS_2]
- Millerite [NiS]

Sulpho-salts
- Enargite ✓(5) [Cu_3 As S_4]
- Luzonite ✓(6) [Cu_3 As S_4]

Haloids
- Fluorite[3] [CaF_2]

Oxides
- Chert (Jasperoid) ✓(11) [SiO_2]
- Quartz ✓(12) [SiO_2]

Carbonates
- Dolomite ✓(9) [Ca Mg $(CO_3)_2$]
- Calcite ✓(10) [$CaCO_3$]
- Aragonite [Orthorhombic form of $CaCO_3$]

Phosphate
- Apatite ✓(13) [$Ca_5(PO_4)_3F$]

Silicates
- Kaolinite ✓(14) [$2H_2O \cdot Al_2O_3 \cdot 2SiO_2$]
- Glauconite[4] $K_2(Mg,Fe)_2Al_6(Si_4O_{10})_3$-$OH_{12}$

Sulfates
- Barite[4] ✓(15) [$BaSO_4$]
- Gypsum ✓(16) [$CaSO_4 \cdot 2H_2O$]

Miscellaneous
- Petroleum [Liquid hydrocarbons]
- Bitumen [Solids & Semi-Solid hydrocarbons]

See Table 5 for ore zones in Boone Formation (at end of text)

SECONDARY MINERALS

Elements
- Sulphur ✓(6) [*elemental* S]

Sulfides
- Greenockite [CdS]
- Wurtzite [ZnS]
- Covellite [CuS]

Oxides
- Limonite [FeO (OH) · nH2O]
- Hematite [Fe_2O_3]
- Pyrolusite [MnO_2]
- **Cuprite [Cu_2O]**

Carbonates
- Smithsonite ✓(1) [$ZnCO_3$, dry bone]
- Cerussite ✓(4) [$PbCO_3$]
- Hydrozincite [$2\ ZnCO_3 \cdot 3\ Zn(OH)_2$]
- Malachite ✓(5) [$Cu_2(OH)_2$ CO3]
- Azurite [$Cu_3(CO3)_2(OH)_2$]
- Aurichalcite [$2(Zn, Cu)CO_3 \cdot 3(Zn, Cu)(OH)_2$]

Phosphates
- Diadochite ✓(14) [$Fe_2(PO_4)(SO_4)(OH) \cdot 5H_2O$]
- Pyromorphite [$Pb_5(PO_4,AsO_4)_3Cl$]

Silicates
- Calamine ✓(2) [$(ZnOH)_2 SiO_3$]
- Chrysocolla [$CuSiO_3 \cdot 2H_2O$]
- Allophane [colloidal hydrous Al silicate clay mineral – [$SiO_2 \cdot Al_2O_3 \cdot 2H_2O$])

Sulfates
- Anglesite ✓(3) [$PbSO_4$]
- Goslarite ✓(7) [ZnSO4 · $7H_2O$]
- Epsomite ✓(8) [MgSO4 · $7H_2O$]
- Gypsum ✓(9) [$CaSO_4 \cdot 2H_2O$]
- Melanterite ✓(10) [$FeSO_4 \cdot 2H_2O$]
- Szomolnokite ✓(11) [$FeSO_4 \cdot H_2O$]
- Copiapite ✓(12) [Fe, Mg $Fe_4(SO_4)_6(OH)_2 \cdot 20\ H_2O$]
- Carphosiderite ✓(13) [$H_2O \cdot Fe_3(SO4)_2(OH)_5 \cdot H_2O$]
- Barite [$BaSO_4$]
- Leadhillite [$Pb_4(SO_4)(CO_3)_2(OH)_2$]
- Caledonite [$Cu_2Pb_5(SO_4)_3(CO_3)(OH)_6$]
- Linarite [$PbCu(SO_4)(OH)_2$]
- Chalcanthite [$CuSO_4 \cdot 5H_2O$]

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[3] Fluorine present, but combination uncertain. ? – presence questionable
[4] Present but possibly secondary
✓From McKnight & Fischer, 1970, pp. 102-124 (# in parenthesis indicates order of importance).

Contamination Generation

- ✓ Acidity – Pyrite & Marcasite Weathering
- ✓ Metal ions – Sulfide Mineral Weathering

Attenuation Processes

- ✓ Alkalinity from calcite, dolomite and aluminosilicate weathering
- ✓ Precipitation of metal ions
- ✓ Sedimentation of Ochre (pH dependent forms, see text)
- ✓ Sorption of metal ions

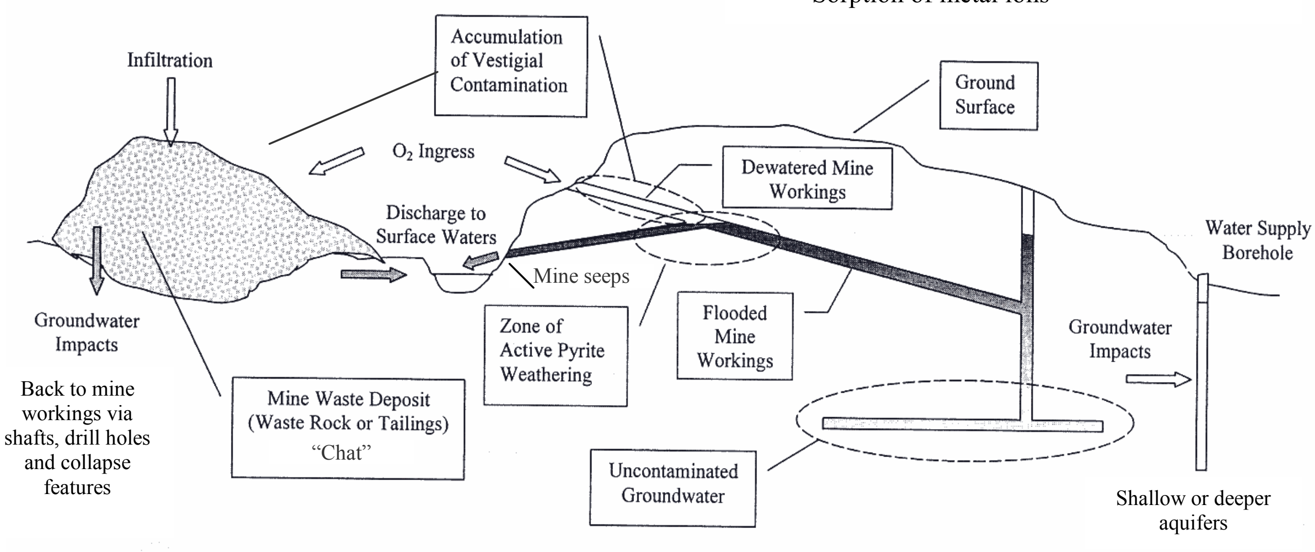

Figure 6. Diagram of mine workings where oxidative weathering of sulfide minerals produces acidity, metal ions and ochre (iron salts, see text) as significant environmental hazards via transport through surface water and groundwater to sensitive receptors such as abstraction points for surface water (streams & ponds), groundwater wells (boreholes) and stream biota. Active (*juvenile*) weathering occurs where oxygen-rich (oxic) waters are in contact with sulfide-bearing rock (see Table 4). In the absence of active transport of contaminants, secondary (*vestigial*) weathering products can accumulate contaminants as solid precipitates in hydraulically unsaturated zones within surface workings or around dry or partially wetted mine shafts, fractures, collapse features or unsaturated upper mine levels. Attenuation processes produce buffering solutions or generate precipitates or sorption surfaces that reduce acid mine effects but may not eliminate them completely. [Modified from Younger et al, 2004].

during these reactions can be used by bacteria as an energy source to grow and to maintain cell functions. In the majority of situations, microbial activity plays a

significant role in how fast these redox reactions occur. Detailed models of the reaction mechanism and reaction rates generally must consider a number of linked geochemical and biochemical reaction steps (Younger, et al., 2002).

The reactions listed below only represent an average reaction geochemistry of overall weathering processes. Such geochemical descriptions are useful for assessing contaminant mass balances but provide no information on factors that influence weathering rates.

Pyrite weathering that produces sulfate (sulfuric acid) also releases soluble ferrous iron (Fe^{2+}) and acidity which is represented by production of protons in equation (1). The following reactions are also relevant for the weathering of the mineral marcasite, which has the same composition of pyrite (FeS_2(s), but with a different crystallographic structure.

$$2FeS_2(s) + 7O_2(aq) + 2H_2O \sim 2Fe^{2+} + 4SO_4^{2-} + 4H^+ \qquad (1)$$

If sufficient water dissolved oxygen is present, or if solutions can be oxygenated by contact with the atmosphere, the dissolved ferrous iron (Fe^{2+}) will be oxidized to ferric iron (Fe^{3+}), consuming acidity in the process.

$$2Fe^{2+} + ½\, O_2 + 2H^+ \rightarrow 2Fe^{3+} + H_2O \qquad (2)$$

A much greater net production of acidity occurs because ferric iron can *react further* to precipitate as iron oxyhydroxide forming ochre (Eq. 3 below), or by reacting further with the pyrite (or marcasite) to produce more acidity and ferrous iron (equation 4). Because both the forward and reverse reactions are relatively fast for the precipitation and dissolution of ferric hydroxide, compared to the residence times of discharge water in the mine workings, mine waters often achieve solubility equilibrium with these minerals. This is denoted by writing Equation (3) as both a forward and reverse reaction at equilibrium.

$$2Fe^{3+} + 6H_2O \leftrightarrow 2Fe(OH)_3(s) + 6H^+ \qquad (3)$$

$$14Fe^{3+} + \underbrace{FeS_2(s)\ \}}_{\text{pyrite or marcasite}} + 8H_2O \rightarrow \underbrace{2SO_4^{2-} + 15Fe^{2+}}_{\text{Produces acidity}} + 16H^+ \qquad (4)$$

The ferrous iron produced in reaction (4) can then be re-oxidized by available dissolved oxygen, perpetuating the cycle represented by reactions (2) – (4). If dissolved oxygen becomes used up (depleted), reaction (4) can proceed to completion yielding predominantly ferrous iron in solution.

Metal sulfides other than pyrite will not necessarily produce acidity, but will release soluble metal ions to solution. Some examples are given below where the corresponding sulfide mineral name is listed to the left of each weathering reaction.

Sphalerite	$ZnS(s) + 2\, O_2(aq) \rightarrow Zn^{2+} + SO_4^{2-}$	(5)
Galena	$PbS(s) + 2\, O_2(aq) \rightarrow Pb^{2+} + SO_4^{2-}$	(6)
Millerite	$NiS(s) + 2\, O_2(aq) \rightarrow Ni^{2+} + SO_4^{2-}$	(7)
Greenockite	$CdS(s) + 2\, O_2(aq) \rightarrow Cd^{2+} + SO_4^{2-}$	(8)
Covellite	$CuS(s) + 2\, O_2(aq)) \rightarrow Cu^{2+} + SO_4^{2-}$	(9)
Chalcopyrite	$CuFeS_2(s) + 4\, O_2(aq) \rightarrow Cu^{2+} + Fe^{2+} + 2\, SO_4^{2-}$	(10)

It is clear from these weathering reactions that water infiltrating mine workings will accumulate solutes as minerals dissolve. As acidity is released to solution during pyrite weathering, the pH will drop. There are other minerals (calcite, dolomite, chert, etc.) however, that can consume acidity by producing alkalinity as they dissolve, thereby providing natural attenuation of the acidity produced by pyrite weathering, helping to buffer the pH. This is a critical aspect of assessing mine water contamination since the solubility and therefore the mobility and bioavailability of metal ions is strongly pH dependent. Metal ions generally have increasing solubility with decreasing pH. Natural attenuation of acidity and pH buffering are the critical controls on the environmental behavior of metal ions that are released in mine water discharges (Evangelou, 1995; Younger et al, 2002).

Acidity produced by active pyrite weathering transported by advective flow from the zone of weathering is termed *juvenile* acidity. It is called this because the combination of active weathering and transport will cause recently produced AMD contamination to develop relatively quickly where the flow emerges from the mine workings. In Figure 6, the active zone of pyrite oxidation at the water table is an example of where juvenile acidity is produced.

Vestigial acidity develops in hydraulically unsaturated void spaces that are in contact with the atmosphere (oxygen source) but not actively flushed by water flow (wet then dry precipitation, Younger,1997a). This void or pore space includes the vertical and horizontal walls and hydraulically unsaturated rock around dewatered underground workings, and the pore space in mine waste deposits situated above the water table. Because these voids are not actively flushed, sulfide weathering products accumulate within the pore waters; acidity as H^+ and as metal ions such as aluminum (Al^{3+}) and ferric iron (Fe^{3+}). When wetter conditions prevail these acidic salts can be washed into the surface environment.

Under persistently oxic conditions that drive weathering of pyrite and other sulfide minerals, metal ions and sulfate will continue to accumulate in pore waters until the solubility limit for metal sulfate and hydroxy-sulfate minerals is reached. As sulfide weathering continues, these secondary weathering products accumulate as mineral precipitates. These accumulated precipitates thus represent stored or vestigial acidity which is released to solution only upon cessation of dewatering or upon flooding or flushing of a waste deposit (Younger, 2002; Evangelou, 1995; USPHS, 1940) .

Contact with inflowing groundwater during mine water rebound, after dewatering is stopped, will provide dilution water that flushes the voids causing these secondary phases to dissolve and become available for mobilization. This releases Al^{3+} and Fe^{3+} to solution where they contribute significantly to the acidity load in the rising waters. In addition, H^+ that has accumulated in the previously immobile void water will also contribute to the acidity. Because dissolution of many of these secondary hydroxysulfate minerals is rapid compared to hydraulic transport during mine water rebound, their constituent ions will accumulate progressively as mine waters rise, resulting in a potentially catastrophic release of contamination when the discharge emerges. The contamination will then subsequently subside over time as the workings are continually flushed by recharge water and the rise and fall of the water level continues dissolving and depositing these salts upon fall of the water level and remobilization upon rewetting. When the vestigial acidity is flushed from the system, any zones of active pyrite oxidation will continue to produce juvenile acidity until pyrite is exhausted (depleted). This process can take decades to centuries depending on pyrite availability and lack of offsetting by natural alkalinity.

Impact of Biota on Acidity and Alkalinity

In addition to oxidative pyrite weathering, as well as the weathering of carbonate and silicate minerals, there are a number of other processes which can influence acidity and alkalinity of natural waters. Figure 7, compiled from Younger *et a*l (2002) and Schnoor and Stumm (1985), lists a number of these processes, and represents them as quantitative reactions showing uptake or release of protons (H^+) as a sink or source for acidity. Anaerobic respiration can be particularly important in wetlands and stream sediments due to the presence of detrital organic carbon consuming oxygen (O_2) by aerobic respiration at the water-sediment interface. In underlying anaerobic sediments, microbial sulfate reduction by organic carbon can occur, consuming acidity and providing sulfide ions (S^{2-}) which can remove metal ions by forming new sulfide minerals (FeS(s), ZnS(s), etc). These sulfur reduction process reverses the effects of sulfide mineral oxidation that occurs in oxic environments, thus providing natural attenuation of acidity and metals contamination.

Processes involving biomass growth and dieback can have a large influence on soil chemistry. This may occur for example during reclamation of abandoned mine sites through composting or establishment of vegetation covers. Figure 7 represents biomass growth as uptake of nutrient ions and their incorporation into plant tissue. The predominance of base cations as nutrients means that biomass growth extracts alkalinity from the soil.

According to the definition of strong acids and bases[5] (see Butler, 1998, p. 70), loss of base cations corresponds to loss of strong base or a decrease in alkalinity; i.e. acidification of the soil. Decomposition of organic matter after dieback returns these nutrients and the associated alkalinity to the soil. Composting on cover layers thus returns alkalinity, stored in the biomass, to the soil (Younger *et al*, 2002; Naftz *et al*, 2002).

Any biological or chemical processes that influences the charge balance of natural waters; either through proton consumption or release by redox processes or uptake and release of either cations or anions, will generally influence the acidity and pH. In mine waters, sulfate is a tracer for input of strong acid by pyrite weathering while base cations are tracers for inputs of strong bases by carbonate or silicate weathering or other attenuation processes. **In summary, "processes that produce or consume the cations**

[5] The term *strong* and *weak* refer not to the concentration of a solution but to the extent to which a substance is dissociated to its ions in solution. A strong electrolyte is one that is completely dissociated, and weak electro lye is one that is only partly dissociated. "Strength" depends on the solvent as well as the solute. HCI is completely dissociated in water, because water is an excellent proton acceptor, but pure (glacial) acetic acid is a poor proton acceptor and HCI is a weak acid in that solvent. Similarly, alkali metal hydroxides (such as NaOH, a strong base) are fully dissociated (hence "strong") in dilute to moderate concentrations, but are somewhat associated (hence "weak") at high concentrations (above1 molar).

Figure 7. The influence of biomass and anaerobic processes on the proton (H^+)balance of mine waters (after Schnoor and Stumm, 1985; adapted from Younger et al (2002)).

Biomass production and decomposition produce and consume acidity

Nutrients		*biomass*	*acidity and O_2 release*
$800\,CO_2$		$(CH_2O)_{800}$	
$6NH_4^+$		$(NH_3)_8$	
$4Ca^{2+}$	production	(H_3PO_4)	
$1Mg^{2+}$	$\longrightarrow$	(H_2SO_4)	
$2K^+$		$(CaO)_4$	$+16H^+ + 8040_{2(g)}$
$1Al(OH)_2^+$		(MgO)	
$1Fe^{2+}$		$(Al(OH)_3)$	
$2NO_3^-$	$\longleftarrow$	(FeO)	
$1H_2PO^{4-}$	decomposition	$(H_2O)_m$	
$1SO_4^{2-}$			

Aerobic respiration and fermentation reactions do not affect acidity

$$O_{2(g)} + CH_2O \rightarrow CO_{2(g)} + H_2O$$
$$2CH_2O \rightarrow CO_{2(g)} + CH_{4(g)}$$

Anaerobic respiration

$$5/4CH_2O + 1/4NO_3^- + H^+ \rightarrow 5/4CO_{2(g)} + 1/2N_{2(g)} + 7/4H_2O$$
$$CH_2O + 2MnO_{2(s)} + 4H^+ \rightarrow 2Mn^{2+} + CO_{2(g)} + 2H_2O$$
$$CH_2O + 4Fe(OH)_{3(s)} + 8H^+ \rightarrow 4Fe^{2+} + CO_{2(g)} + 11H_2O$$
$$CH_2O + 1/2SO_4^{2-} + H^+ \rightarrow CO_{2(g)} + H_2S_{(g)} + H_2O$$

Anaerobic oxidation of pyrite

$$2NO_3^- + FeS_{2(s)} + H^+ \rightarrow N_{2(g)} + SO_4^{2-} + Fe^{2+}$$

of strong bases or the anions of strong acids will alter the acidity, alkalinity and pH of a mine water" (Younger *et al.*, 2002)**.**

Younger *et al.* (2002, pp. 98-112) shows that the composition of mine water can be used together with the discharge flow rates to determine solute fluxes. These fluxes can be related to the rates of key processes that generate and attenuate contamination loads. If the mass of contaminant source minerals can be estimated, these rates can be used to estimate the contamination lifetime of a mine site, and the time to depletion for sources of natural attenuation. With such information, the basic concepts of acidity, alkalinity and pH buffering can be extended to assess trends in discharge water quality over time.

Contaminate Load Assessment

Both hydrological and water quality data are needed to perform even preliminary environmental assessments of mine sites. Because the critical contaminates are solutes (chemical forms in solution), water flows (litres/day (L/s) or cubic feet/second (cfs)) and ion concentrations (mole/litres (M/L) or millimoles per litre (mM/L) are required in order to determine contaminant fluxes (mile per day).

Below, initial attempts taking known flow rates for various reaches of Tar Creek and combining them with appropriate water quality data to estimate order-of-magnitude solute fluxes, contamination loads and the rates of processes that generate and attenuate or reduce the contamination.

Finally, methods for determining the potential for mine contamination have been developed by the U.S. EPA as a standard method of *Acid Base Accounting* (ABA, see Sobek *et al.*, 1978; Kleinmann, 2000) for pre-mine assessment of pollution potential and site monitoring requirements.[6] A number of shortcomings and subsequent modifications to the procedure have been developed (see Li, 1997; Kwong & Ferguson, 1997; Lawrence & Wang, 1997). Additional assays have been developed. These include measuring the Net Acid Generation (NAG) by reacting rock with hydrogen peroxide at room temperature until available mineral sulfide is chemically oxidized (see Finkelman & Griffin, 1986; O'Shay *et al.*, 1990; ASTM, 1996(2001); Price 1997).

Another type of test called *kinetic tests* include laboratory batch column reactors containing mine rock in contact with aqueous solutions, where solute concentrations are followed with time. The tests attempt to extrapolate observed weathering rates for acid-forming and acid neutralizing minerals, and the associated impact on site discharge quality, to field conditions over the lifetime of a mine. Large differences in the physical scale and complexities between lab tests and field sites make such predictive trends very uncertain. Kinetic tests have also been undertaken *in-situ* at mine sites to reduce the scale dependence of the test.

At present, current state of the art for conducting static and kinetic tests for surface coal mining has been summarized by Kleinmann (2000). No similar consensus exists for applying such tests to hard rock mining environments. Such methods are nevertheless being pursued and recent research suggests methods will soon be developed to reliably extrapolate weathering-rate test data to field conditions in the hard rock context. For example see Malmström et al, (1999) study where the weathering rates determined from

[6] The method compares the acid-neutralizing capacity, termed Neutralization Potential (NP), and the acid-producing capacity, termed Acidity Potential (AP), in order to determine if a particular sample of material is expected to produce acidic effluents when weathered. The NP is determined in a laboratory assay where a fixed mass of crushed rock is reacted with a known excess of concentrated hydrochloric acid while heating. initial acidity in order to determine the amount of protons consumed. The NP is generally reported in equivalents of calcite per kilogram of rock that have reacted with the hydrochloric acid ($CaCO_3(s) + 2H^+ \rightarrow Ca^{2+} + H_2O + CO_2g$)). The AP is determined by chemical analysis of the rock, relating total sulfur content to equivalents of sulfuric acid (H2SO4) that could be produced from sulfide weathering. The AP, is likewise expressed as equivalents of calcite per kilogram of rock. The Net Neutralization Potential (NNP) is the difference between the two: i.e., NNP = NP - AP (as g CaCO3(s)/ Kg rock).

kinetic tests using batch (0.15 Kg rock sample), large column (1820 Kg or 2 short tons) and field (9.0 x 10^{10} Kg or 99 million short tons) studies were compared for a single waste rock deposit. Results showed that the scale-dependence observed the site being studied is mostly predictable by quantifying the effects of temperature, reduced oxygen content, particle-size distribution, spatial variability in mineralogy and the content of stored (immobile) water at the site (see Banwart *et al*., 1998).

Sulfide weathering rates are measured at field scale in order to monitor consumption of O_2 in isolated sections of tailings deposits (Blowes *et al.*, 1991) or in packed-off (isolated) sections of boreholes drilled into mine rock (Harries and Ritchie, 1985). Detailed monitoring of site hydrochemistry can be used to supplement the methods outlined here. This includes gaging discharge flows and monitoring water quality. Lysimeters can be installed in the field in order to sample water flows in hydraulically unsaturated mine rock (Ericksson *et al.*, 1997). Sampling from borehole sections located below the water table can be used to monitor groundwater quality.

Some of these data collection methods have been done at Tar Creek in very limited applications over limited periods but in no systematic or consistent fashion as suggested here.

Current Hydrologic System – Tar Creek Watershed

General Features

In terms of general hydrologic features and processes, many reports have been published regarding the system of flow and water quality. These include but are not limited to: OKOSE (2004), USEPA (2000, 2004), Governor Keating's Tar Creek Superfund Task Force Final Report (2000), Luza (1986), Marcher and Kenny *et al.* (1984), Marcher and Bingham (1971), McKnight and Fischer (1970) and Reed *et al.* (1955). Mining and geologic perspectives are given by Bastin (1939, Behre et al. (1950), Fowler (1942), Fowler and Lyden (1932), Fowler (1935), Newhouse (1933), Ohle (1958), Ridge (1936), Snyder (1966), Snyder and Gerdemann (1968) and Stoiber (1946).

These reports provide the overall best perspective of the watershed in context to the mining activities as well as other salient hydrologic and geologic features, quality or processes that can be used to develop a more comprehensive appreciation for watershed management planning purposes. This information, together with stream flow information collected previously by the OWRB (1989), and USGS (1993) regarding gaging and surface water quality field & sampling data and various specialty reports: ODEQ (2004), OWRB (1982; 1983a,b); Playton *et al.* (1980); Parkhurst (1986), Parkhurst et al., (1988), Christenson (1995); DeHay (2003); and more general information regarding surface flows in smaller draining creeks are found in Havens (1978); Lurry & Tortorelli (1995); Tortorelli (1997).

There are numerous other reports and special studies not listed here that were previewed who's results were made part of other later studies but they were not used directly used for the present evaluation due to time and availability constraints. What is important however is that each new and/or existing information source should be utilized to refine the hydrologic system function (concept) perspective and future efforts to define data, information or design investigative requirements should become integrated into the master concept of the hydrologic functioning of the Watershed integral with the mine workings flow system. This is the motivation for the USGS modeling being undertaken as part of the watershed management plan work.

Future management options, alternatives and key decision-making will be dependent on this consistent and monitored perspective. This validates the point that committing to regularly monitoring the water resource at known, preferably fixed locations throughout the watershed, as remedial and other watershed-modifying activities progress becomes essential for comparative analysis of remedial project effectiveness in creating improved hydrologic responses within the watershed. Indeed, the Congressional motivations for all improvements of assessing, defining and remediation planning to mitigate environmental impacts from all forms of coal mining provided by the SMCRA (1977) forms the basis of the improved environmental health of those areas today. More on this regarding the water resources monitoring within the Tar Creek Watershed is provided in the section below regarding "*Framework Monitoring Requirements (Networks).*"

General hydrologic characteristics of the region including the Tar Creek Watershed and Lower Spring River Watersheds in the wider context as tributaries in the Neosho River Basin is provided by Marcher and Kenny *et al.* (1984, Area 40). Another, more recent, but different emphasis and development perspective on impacts to Grand Lake (similar area but more eastern portion Area 40) found in OKOSE (2004). Both reports cover a much larger area than the watersheds currently being considered, their region extending much farther to the west but includes all the area of the present focus, even those portions of Spring River into Kansas and Missouri. General features summarized by Marcher & Kenny *et al.* include geology, physiography, surface drainage, soils, land use and climate (Marcher and Kenny et al., 1984, p. 20-32) as well as the more specific surface and ground water resources (summarized subsequently below). Relevant portions of their report are summarized here as a basis of conditions for future as well as further analysis in this planning report. More recent interpretations specific to the mining hydrology within Tar Creek will be given subsequent to this.

Marcher and Kenny et al's report covers what is known as Area 40 of the Western Region, Interior Coal Province that covers portions of southwestern Missouri, southeastern Kansas and northeastern Oklahoma including Tar Creek and the entire reach of the Spring River (see Figure 8). There stop-unit format consisting of a brief text accompanied by a map, graph, table, or other illustration is commendable and should be adapted as a quick way to convey multiple sources and types of information useful in the watershed management process. A summary of this information regarding the general geology, physiography, surface drainage, soils, land use and climate for Area 40 follows (Marcher & Kenny et al., p. 20-32)

> Area 40 encompasses about 16,300 square miles in the Arkansas River basin. Principal drainage systems within the area are the Verdigris and Neosho Rivers, which drain predominantly shale, sandstone, and limestone of Pennsylvanian age, and the Spring River,

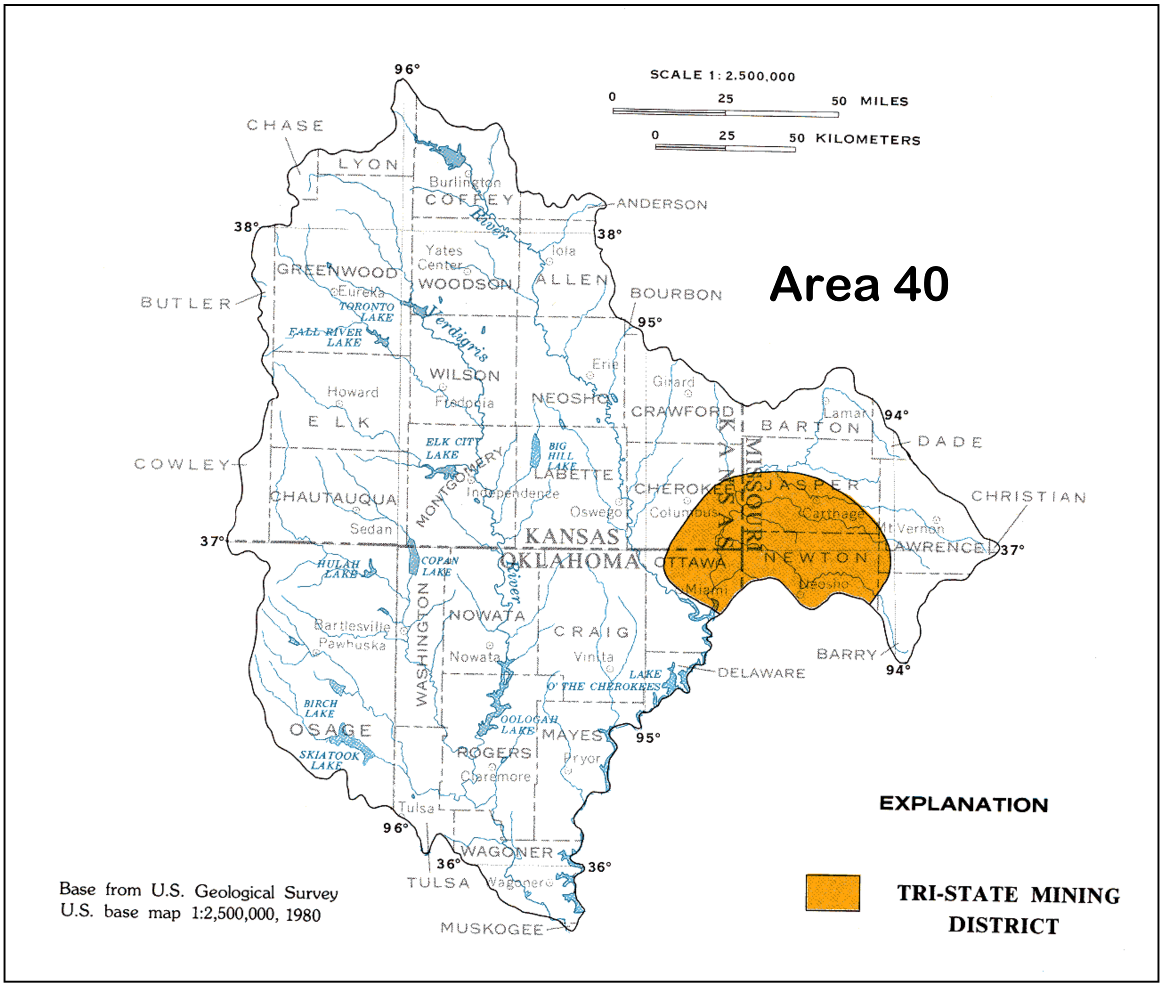

Figure 8. Location of the Tri-State zinc and lead mining district within Area 40 (modified from Feder et al., 1969 by Marcher and Kenny, et al., 1984, Fig. 1.6-1).

which drains mostly cherty limestone of Mississippian age. Pennsylvanian rocks in certain parts of Area 40 contain coal reserves of potential economic importance. Many of the smaller drainage basins have been affected by past and current coal mining, as well as petroleum production and zinc and lead mining. Land use and land cover are determined by the physiographic and soils characteristics of the area. In the Osage Plains [Oklahoma & Kansas] where silt and clay loams predominate and relief is low, most of the land is nonirrigated cropland, managed pasture, or rangeland covered with native grasses. The steeper, stonier soils of the Springfield Plateau [Missouri] are covered with a greater proportion of woodlands.

Mean annual rainfall ranges from about 32 inches in the western part of the area to about 42 inches in the eastern. Because wells in most of the area yield only small quantities of water suitable for domestic and stock use, the sparse population relies principally on surface-water sources for public water supplies and industrial use. About 25 percent of the population is served by rural water systems. Lakes and reservoirs are an integral part of the surface-water drainage system in the area, minimizing floods and providing water supplies. The U.S. Geological Survey has collected systematic data at 166 surface-water stations in Area 40, of which 72 are currently active [circa 1984, pending 2004 update]. Available data

> include records of stage, discharge, and water quality of streams and records of stage and contents of lakes and reservoirs. Additional data have been collected at about 335 miscellaneous surface-water stations, many of which are in mined areas. Records from 91 stream-water-quality stations indicate great variability in pH and in concentrations of iron, manganese, and dissolved solids with the most mineralized water in areas previously disturbed by resource-recovery activities.
>
> Water-level data are available for 438 wells in Area 40; water-quality data are available for 516 wells and springs. The greatest yields of useable ground water are obtained from wells in the unconsolidated deposits of stream valleys in Kansas and Oklahoma and from deeper rocks of the Cambrian and Ordovician Systems in the eastern part of the area.

Hydrologic data for Area 40 are stored in computer files accessible through the USGS National Water Information System (NWIS) website at http://waterdata.usgs.gov .

Salient technical points to remember in considering this area regarding past coal and metal mining impacts (Marcher and Kenny, et al., p. 6, 12-39) include:

1) Approximately 430 million tons of coal have been mined by underground and surface methods in Area 40. Coal operations have resulted in about 75,000 acres of unreclaimed land, most of which are not affected by reclamation laws.
2) Ponds left when coal mining was completed can provide valuable surface-water storage. Mine ponds provide habitat for wildlife and may be a water-supply source if the quality is suitable.
3) Mining may increase storage of ground water and decrease peak discharges of streams. Reclamation practices that improve the cover of vegetation on mine spoil may decrease the volume of precipitation that runs off quickly thereby allowing the water to infiltrate the spoil where it is stored and gradually discharged to streams.
4) Wastes from oil fields and refineries have adversely impacted some areas in the western half of Area 40. Large concentrations of dissolved solids, particularly chloride, are common in surface and ground waters degraded by disposal of brines and wastes.
5) Regarding the Tri-State mining area: Mining of zinc and lead has degraded the chemical quality of some surface and ground waters in the eastern part of Area 40.
6) Geologic structure of Area 40 is dominated by the Ozark uplift, a broad dome centered in southern Missouri and extending into southeastern Kansas and northeastern Oklahoma. The area is on the western flank of this structural high meaning progressively younger age rocks outcrop from east (Mississippian age) to west (Pennsylvanian age). Regional dip is toward the west-northwest about 30 feet per mile (ft/mi) interrupted by folds of small magnitude and minor faults with displacements on tens of feet to larger features with displacements of several hundreds of feet. Rocks of Mississippian age are mostly cherty limestone; rocks of Pennsylvanian age are mostly shale with some limestone, sandstone and coal.
7) Area 40 includes parts of two physiographic provinces: Missouri portion is in the Springfield Plateau section of the Ozark Plateaus Province, nearly all the

area in Kansas and Oklahoma is in the Osage Plains Section of the Central Lowlands Province.

8) Springfield Plateau in the east is carved on cherty limestone of Mississippian age and has relatively uniform resistance to erosion. As a result, land surface along the broad interstream divides is gently rolling with local relief of about 50 feet. Away from divides and toward streams, the topography becomes hilly to rugged with local relief ranging from 100-200 feet. Larger valleys are flat-bottomed and steep-walled; tributary valleys general have sharp v-shaped cross-sections.
9) The Springfield Plateau merges without notice into the Osage Plains lying to the west. Immediately adjacent to the Plateau is a belt of 20-25 miles wide underlain by shales of the Krebs and Cherokee Groups. These are weakly resistant rocks having weather and eroded to form broad, nearly featureless plain with maximum relief of approximately 50 feet.
10) Soil characteristics in Area 40 vary with geology and physiography. Most soils were developed from shales, are acidic, and have low to moderate fertility, very slow to moderate permeability, and moderate erosion potential.
11) Most of Area 40 consists of Farms and Rangelands. Land-use and land-cover maps and resource evaluation using remote-sensing techniques provide detailed information for many applications. The immediate area within the Tri-State Pb-Zn Mining District consists predominately of woodland, and roughly equal proportions of cropland and rangeland or pasture.
12) Area 40 is characterized by a humid, continental climate. Average annual precipitation varies from 32 along the western edge to 42 inches along the eastern edge of the Area; late summer to early winter being usually the driest period and spring is usually the wettest. A large percentage of rainfall occurs during short, intense thunderstorms associated with squall lines ahead of frontal systems during spring and summer with about 75% of the annual precipitation falling during this period. The 10-yr, 24-hr rainfall event varies from 5.5 inches in the north to 6.5 inches along the southern portions of the Area.
13) Mean temperatures range from 56º F in the north to 61º F in the south. Growing season ranges from 190 days in the north to 210 days in the south.
14) Annual frequency percentage for wind direction and velocity are fairly uniform; prevailing southerly winds occur during all months except January and February when they are primarily from the north. Average annual wind speeds are 10-12 miles per hour. Storm gusts associated with squalls and fronts are usually 30-40 miles per hour associated with summer thunderstorms and cold fronts in winter..
15) Due to the warm temperatures and windy conditions, evaporation is significant. Average pan evaporation ranges from 60 (east) to 75 inches (west).
16) Principle use of water in the area is for public supply (1980). While surface water predominate water use in Kansas and Oklahoma; Missouri's water use is approximately equal volumes of surface and ground water.
17) About one-fourth of population supplied with water from rural water systems. Of 140 rural water districts in the area, 118 derive supplies from surface-water sources and 22 from ground water systems (1980 accounting).

Hydrologic characteristics are as follows:

1) Streamflow varies seasonally with minimum streamflow occurring in late summer and autumn with many streams ceasing flow during this period (streams generally with basin sizes < 250 sq mi can cease flowing if not spring fed). Maximum streamflow usually occurs in Spring when rainfall is greater and before evaporation reaches its summer peak.
2) Streamflow data from gaging stations having 20 or more years record show an average annual water yield of 0.57 cubic foot per second (cfs) in the Neosho Basin (includes Tar Creek Watershed) and an average annual water yield of 0.79 cfs in the Spring River basin. These differences largely reflect the water-storing characteristics of the geologic rock underlying each basin, respectively.
3) The Neosho basin is underlain mainly by shale and sandstone that have limited capacity to store water. The basins within the Spring River is underlain by weathered chert and limestone that can store a large volume of water that is released slowly to the river and its tributaries. Mean annual runoff and flood-discharge characteristics with selected recurrence intervals can be estimated. Mean annual runoff and flood-discharge characteristics with selected recurrence intervals can be estimated for perennial, intermittent, and ephemeral streams where gaged stream flows are not available by use of equations developed for this purpose (Hedman and Osterkamp, 1982). These empirical equations permit the use of channel-geometry measurements as an alternative method of quickly estimating streamflow characteristics for ungaged streams.
4) Duration of streamflow is affected by basin characteristics and man's activities. Flow duration information is the distribution of stream discharge with time and is useful for various hydrologic analyses such as determining the yield of a stream or its ability to assimilate waste. Flow duration is determined from flow-duration curves showing the percentage of time that a particular discharge at a given point is equaled or exceeded. For example, the discharge of Shoal Creek above Joplin, Missouri, equaled or exceeded 0.62 cubic foot per second per square mile 40 percent of the time (Marcher & Kenny et al, 1984, see fig. 5.2-1). Multiplying the unit discharge value (0.62) by the area of the drainage basin (410 square miles) shows that Shoal Creek discharged 254 cubic feet per second or more 40 percent of the time.
    a. Streams in the Neosho basin flow about 80% of the time; streams in the Spring River basin flow all or nearly all the time. Differences in duration of flow are a result of the differences in geology. Pennsylvanian formations underlying the Neosho basin are mostly shale and sandstone that have little capacity to store and transmit water. On the other hand, Mississippian formations underlying the Spring River basin store and transmit water readily to the streams thereby maintaining flow during dry weather.
    b. Flow-duration curves are useful for relating streamflow to the physical characteristics of a basin. The upper end of the curve shows the direct-runoff characteristics, which are affected by climate, topography, and land use. The lower end of the curve shows the base-flow characteristics, which usually depend on the capacity of rock formations underlying the basin to

store and transmit water. A steep slope of the lower part of the curve indicates that the volume of ground water available to sustain streamflow during dry weather is limited. A flat slope of the lower part of the curve indicates that streamflow is sustained during dry weather by ground-water discharge, addition of municipal or industrial wastes, or reservoir regulation.

   c. The flow duration response of a basin can be significantly changed by man's activities. For example, the volume of direct runoff may be increased by urbanization or decreased by impoundment. Stream reaches "leaking" to the subsurface through open holes, shafts, and collapse features as occurs at Tar Creek can also impact it. Additionally, base flow may be augmented by discharge of municipal or industrial wastes or depleted by consumptive use (as occurs by sewage flows to Lytle Creek. If such changes are great enough, the shape of the flow-duration curve will reflect them.
   d. Flow-duration data for gaged streams in Area 40 are given in reports by Furness (1959, 1960) for Kansas, Skelton (1976, 1977) for Missouri, and Mize (1975) for Oklahoma (these may have more current updates).

5) Flooding is most likely to occur between March and July and is least likely to occur between November and February. Flooding along small streams is common in Area 40 and is caused by rainfall associated with local, intense thunderstorms in spring and summer. Engineering design for safe and economical structures such as bridges, culverts, embankments, dams, and levees, requires data on the magnitude and frequency of floods. Regulatory guidance from the Surface Mining Act (PL 95-87) refer in particular to the 2-yr, 24-hour and the 10-yr, 24-hr, precipitation events for design of temporary and permanent structures, respectively.
   a. Although rainfall is the primary cause of floods, no exact correlation exists unless specifically addressed for a specific area. Flood magnitudes are related to the physiography of a basin, including land slopes and drainage patterns. Furthermore, land uses such as farming, mining and urbanization also affect flood magnitudes.
   b. Flood-frequency curves for ungaged, unregulated streams in Area 40 can be estimated by use of regional equations developed for this purpose. These equations are not applicable for urbanized basins without modification (Huntzinger, 1978a-c). Flood-frequency equations applicable to those parts of each State included in Area 40 have been developed independently for Kansas (Jordan and Irza, 1975), Missouri (Sandhaus and Skelton, 1968, and Hauth, 1974), and Oklahoma (Sauer, 1974; Thomas and Corley, 1977; Tortorelli, 1997). These equations were developed similarly; however, they differ in context and application (Marcher and Kenny et al., 1984).
6) Surface water storage is vital to Area 40. In most of Area 40, streamflow is so variable and groundwater availability is limited that surface-water storage is necessary for municipal, industrial, irrigation, and domestic water supplies.
7) Water quality data: For area 40, the stream water-quality data available include that collected at 91 stations (update required for Tar Creek-Lower Spring River

area, see Marcher & Kenny et al., fig. 6.1-1) where at least four determinations of specific conductance and pH have been made. Only 29 stations had more than 100 analyses available.

a. Station name and location are given by Marcher & Kenny et al., 1984 (Fig. 6.1-1, and listed in their section 9.3). The water-quality data were collected for various purposes under a variety of cooperative programs with Federal, State, and local agencies. Therefore, the 91 stations do not represent a *network designed to provide uniform data throughout the area that relate directly to coal mining or other specific hydrologic problems.*
b. Analysis of the available data provides a general summary of the chemical quality of water in the area even though the data are not uniform as to quantity, type, or distribution. Those water-quality characteristics or constituents, described in (8) below, that relate directly to coal (or metal) mining include dissolved solids, sulfate, pH, and iron and manganese. To avoid placing overemphasis on extreme values, median values were reported and used by Marcher and Kenny et al. (1984) to describe the chemical quality of the stream water.

8) Surface water quality (summary circa 1984):

a. Dissolved solids: The median dissolved-solids concentrations determined for 85 stream stations ranged from 122 to 3,100 milligrams per liter (mg/l). Of these 85 stations, 63 have median concentrations of 500 mg/l or less which is the maximum contaminant level set for Secondary Drinking Water standard. Only 2 of the 85 stations had median concentrations greater than 1,000 mg/l. Water containing as much as 1,000 mg/l of dissolved solids is considered suitable from growing all non-sensitive and many sensitive crops and also is suitable for most industrial uses, barring objectional concentrations of specific substances.

   (1) Dissolved-solids concentrations in even a single stream vary considerably during low or base flow and are smallest during high flow. Dissolved solids (DS) concentration is inexpensively estimated in the field by measuring the specific conductance (SC) of the water. Pairs of these data can then be fitted to a regression equation relating them. The regression equation for all 85 stream stations (8,861 data pairs) is expressed as:

$$\mathbf{DS = 0.58\ SC + 12} \tag{11}$$

   (2) Equation (11) explains 98% of the variation between dissolved solids and specific conductance. Until this equation is approved upon by reanalysis and updating, it is hereby taken as valid for Area 40.

b. Sulfate: Sulfate is the best indicator of coal-mine or metal mining (of sulfides) in Area 40. Sources include gypsum, and weathering of sulfides, primarily pyrite and marcasite. Median sulfate concentrations in the area range from 6 to 2,050 mg/l. On the basis of taste and laxative effects, the recommended upper limit is 250 mg/l (USEPA Secondary Drinking Water

Standards). Acclimation to sulfate is rapid and many people can drink water as high as 600 mg/l and not experience any laxative effects (Peterson, 1951). Only 23 of 90 stations had over 250 mg/l sulfate concentrations (Marcher & Lenny et al., 1984, Table 6.3-1 and Figure 6.3-1). For stations with median-sulfate concentrations > 1,000 mg/l, the maximum drainage area is 27 square miles (sq mi) in Area 40. Bevans (1980, p. 14) reported for stations in southeastern Kansas, the following relationship exists:

Mean SO_4 conc. = 308 + 28.07 (%DA) (12)

where %DA = percent of Drainage Area Stripped mine for coal

This equation has a correlation coefficient, r = 0.90 and a standard error of the estimate of 260 mg/l. Hence, the equation explains 81% of the variance in sulfate concentration by the percentage of the basin having been stripped mined. Extensive regression correlation analysis was done by Playton et al. (1980) for mine waters in the Picher Field in Oklahoma.

c. Stream water pH: The pH of a solution determines how acid (acidic) or base (alkaline) it is. A pH of 7.0 is neutral water. The farther above 7.0 is more alkaline, and farther below 7.0 the more acidic.

(1) The median pH at 87 of 90 stations ranged from 6.9 to 8.4. pH was found not to be a good indicator of stream contamination by coal-mine drainage in Area 40. Coal mine drainage not unlike Tar creek metal mine drainage is governed by the oxidation of sulfide minerals and subsequent buffering by carbonate and siliceous minerals. This chemistry is the same as explained previously.

(2) However, water in most of the streams in Area 40 has a pH greater than 7.0 (alkaline) because of the buffering effect of the carbonate-bearing sedimentary rocks: limestone and dolomite. The median pH for 3 stations was less than the minimum allowable effluent limit (pH = 6.0, US Office of Surface Mining Reclamation & Enforcement (USOSMRE, 1979). All three of these stations are on small streams directly draining stripped coal mine areas:

Table 6
Small Stream pH for Acid Streams in Area 40
(Reported by Marcher and Kenny et al. (1984, p. 58)

USGS Sta #	Station Name	Drainage Area (sq mi)	Median pH (circa 1979)
53	Little Cherry Cr. Nr West Mineral, KS	34.0	3.9
63	Brush Cr. Nr Weir, KS	29.0	3.9
80	Zinc Mine tailings area storm runoff nr Joplin, MO	0.01	4.1

d. Total-iron (Fe) and total manganese (Mn): Total-iron and total-manganese concentrations less than 1,000 µg/l (McKee and Wolf, 1963, p. 215) are nontoxic to freshwater aquatic life (but can be as low as 100 µg/l for manganese, Nemerow, 1991, p. 231) and are essential to certain physiological functions of aquatic life. Maximum dissolved-iron and manganese concentrations for drinking water are 300 and 50 µg/l, respectively. These standards are primarily based on staining of clothing and plumbing as well as iron imparting a bittersweet taste detectable by some people at concentrations >1-2 mg/l.
   (1) Iron and manganese are common constituents of rocks and soil, and originate from these sources when in water. Other sources include industrial waste, municipal wastes, corroded metals, and acid mine drainage. Although dissolved iron, because of its significant pH dependency, is not always a good indicator of AMD, total iron can be. Total-iron concentration during storm flow from small, unregulated drainages is proportional to the concentration of suspended sediment in the water.
   (2) Median total-iron concentrations ranged from 0 to 8,550 µg/l for Area 40. The median for 1,804 analyses at 52 stream stations is 300 µg/l. The largest median total-iron concentration occurred at station 80 (see Table 6) which received storm runoff from zinc and lead mine tailings (Marcher and Lenny et al, 1984, p. 60).
   (3) Median total-manganese concentrations ranged from 0 to 8,300 µg/l. The median of all 1,301 analyses at 55 stream stations was 216 µg/l. The median concentration at 4 stations (Sta 53, 61, 62, and 63, see Marcher and Kenny et al., Figure 6.5-3 for Sta locations) exceed USOSMRE 30-consecutive-day average effluent standard of 2,000 µg/l. All 4 stations are on small streams (drainage areas between 29 to 170 square miles) draining strip mined areas in Kansas and have large median sulfate concentrations (390 to 830 mg/l) and large median total-iron concentrations (595 to 3,600 µg/l).

e. Sediment yield: Average annual sediment yields range from 50 to 600 tons per square mile of drainage area for Area 40.
   (1) Most of the sediment transported in Area 40 accompanies runoff from thunderstorms of varying intensity during spring and early summer. Runoff as a direct result of rainfall from any particular basin represents the integrated effect of several interrelated conditions including precipitation, geology, soils, topography, vegetation, and land use. This runoff controls the amount and particle size of suspended sediment transported. The instantaneous concentrations of suspended sediment in transport is also dependent on the availability of transportable materials. Agricultural activities, which are a major land use type in the area, is the most areally significant condition.
   (2) The total amount of sediment passing a station per unit area and time is expressed as sediment yield. Sediment yields in the area are lowest on floodplains, where the reduction in average slope and

runoff velocity decreases the erosive capacity of water. In bedrock areas, sediment yields range from about 50 to over 600 tons per square mile per year (see fig. 6.6-1, Marcher and Kenny et al, 1984, p. 63). Yields generally increase in a westerly direction from the Kansas-Missouri state boundary. The Spring River basin in Missouri is estimated to yield between 100 and 200 tons per square mile per year.

(3) Details of sediment yield determinations, contributing conditions and methodologies used are provided by Marcher and Kenny et al. (1984, p. 62).

f. Selected chemical quality data: Selected quality data are available for 35 Coal-mine ponds in Kansas and for 33 coal-mine ponds in Oklahoma. Water in coal-mine ponds commonly contain large concentrations of dissolved sulfate and total and dissolved iron and manganese, and was acidic. Sampling period for these data was from April 1978 to October 1980 for Kansas ponds in Cherokee and Crawford Counties and from May 1976 to November 1981 for Oklahoma ponds, randomly collected. Vertical profiles of specific conductance, temperature, dissolved oxygen, and pH in the mine ponds in Oklahoma, where ponds were greater than 30 ft deep, were stratified. The data for the two states should not be compared as they were part of completely different studies.

Table 7
Summary of Water Quality Data for Coal-mine Ponds, OK & KS
(Modified from Marcher and Kenny et al, 1984, Table 6.7-1)

Parameter	RDWL*	# Anal.	Std Dev.	Median	Min	Max
Kansas (period of sampling: April 1978 to October 1980)						
SC	--	43	898	1,610	73	3,620
pH	--	46	2.07	6.4	2	8.5
Chloride	250	45	5.43	4.5	0.8	33
Sulfate	250	45	633	980	17	2,200
Fe, total	300	45	8,202	440	80	42,000
Mn, total	50	45	15,865	350	30	74,000
Oklahoma (period of sampling: May 1976 to November 1981)						
SC	--	377	897	755	329	3,730
pH	--	336	0.49	7.8	6.3	9.1
Chloride	250	102	52	5.1	0.7	240
Sulfate	250	105	553	260	61	2,300
Fe, total	300	93	1,471	40	10	10,000
Mn, total	50	93	3,337	270	5	16,000

Units: SC – µmhos/cm at 25º C, pH - std units, Chloride & Sulfate - mg/l, Total Fe, Mn - µg/l
* USEPA Secondary Maximum Drinking Water Levels

9) Ground water occurrence, availability & quality: Unconsolidated deposits (alluvium and terrace deposits) yield only limited amounts of water in most of Area 40. Water wells in these deposits may yield as much as 100 gallons per minute (gpm) locally but most wells yield less than 10 gpm; concentrations of dissolved solids in water from these deposits are usually less than 1,000 mg/l.

Recharge to these deposits is from rainfall and they have a large storage capacity. However, silts and clay (fines) content of these deposits is high and retains water through capillary action, hence sand areas, where occurring and sufficient thickness and extent, can be dependable water yielding zones (> 2 gpm). Saturated thickness in these deposits usually less than 10 feet but can range locally upwards of 30 feet. Where saturated thickness is > 20 ft, yields of 100 gpm from individual wells occur. Best method option to recover ground water from these deposits is through use of batteries of wells or infiltration galleries. Water level fluctuation in wells in these deposits reflect volume of water in storage, annual evapotranspiration, intense rains and near-by stream or river stage changes. Water quality for unconsolidated deposits in Area 40 are summarized in Table 8a.

a. Pennsylvania formations water yield and quality (west Area 40): Formations of this age yield only small amounts of water. Wells in Pennsylvanian formations, other than sandstones of the Ada and Vamoosa Formations and the Douglas Group, generally yield less than 5 gpm; concentrations of dissolved solids in well waters range from 100 to 5,000 mg/l.

Table 8a
Summary of Ground Water Quality in Unconsolidated deposits and Pennsylvanian Water-Bearing Formations, Area 40
(Modified from Marcher and Kenny, et al., 1984, Tables 7.1-1 and 7.2-1)

Parameter	Unconsolidated Deposits				Pennsylvanian Formations			
	# Anal	Min	Median	Max	# Anal	Min	Median	Max
Bicarbonate	10	130	340	470	52	11.5	320	1,170
Calcium	10	37	115	170	52	2	60	550
Chloride	19	7	38	454	149	1	39	3,240
Dissolved solids	10	240	410	889	50	83	527	5,110
Iron (µg/l)	17	10	70	34,000	139	10	80	10,000
Magnesium	10	5	14	87	52	0	10	130
Manganese(µg/l)	10	10	32	1,750	53	10	20	61,000
pH (units)	18	6.0	7.1	7.7	143	4.5	7.3	9.0
Potassium	9	0	1	3	45	0	2	46
Sodium	9	10	22	250	52	4.5	76	1,900
Sulfate	19	2	47	3,970	150	1	54	2,300
All units in milligrams per liter (mg/l) unless noted.								

(1) Rocks of Pennsylvanian age, which are present in about 80 percent of Area 40, consist of a repeated sequence of interbedded shale, sandstone, and limestone; the sandstone beds become more numerous and thicker toward the west (down dip). Shale units typically are 50-100 feet thick; sandstone and limestone units generally are 5-20 feet thick. Excluding sandstones in the Ada and Vamoosa Formations and the Douglas Group occurring in the far western portion of Area 40, the Pennsylvanian formations have little primary porosity and their ability to store and transmit water is extremely limited. Consequently, few wells in these formations

yield more than 5 gpm and many yield much less. Deeper wells might yield more water by intercepting more bedding planes or fractures, but below depths of 200-250 feet the water is likely to be too mineralized for most uses.

(2) In the western part of the area, thick sandstones in the Ada and Vamoosa Formations and the Douglas Group may yield as much as 100 gpm.

(3) Because of the geologic structure, the Pennsylvanian rocks are slightly tilted at the surface exposing bedding planes between the rock layers. These bedding planes are the principal avenues or openings for water entry and movement although joints and fractures may be important in some places. Recharge entering the rocks is derived mainly from precipitation falling on the outcrop; small quantities may be derived by seepage from overlying unconsolidated deposits or from streams. Most discharge is by evapotranspiration, although some water is discharged to streams during periods of high water levels. A very small quantity of water is discharged by pumping wells. Water in the zone of weathered rock, which may be as much as 30 feet thick in some areas, generally is unconfined. Water in bedrock below the weathered zone is confined and some wells may flow at times of high water levels. In shallow, unconfined aquifers the slope of the potentiometric surface coincides with the slope of the land surface so the local direction of water movement is toward the streams and down-valley.

b. Mississippian age formations water yield and quality (east Area 40): Water-yielding rocks of Mississippian age in Oklahoma include the "St. Joe", "Reeds Spring", "Keokuk" Formations referred to as the "Boone" Group by Fay et al. (1979) or "Boone Chert" by Marcher and Bingham (1971). Approximately equivalent rocks in Kansas include the Burlington and Keokuk Limestones (Ebanks et al., 1979) and in Missouri they include the Reeds Spring Formation, Elsey Formation, Burlington Limestone, and Keokuk Limestone (Thompson, 1979, and Feder et al., 1969). These formations, which have an combined thickness of 300-400 feet, consist principally of cherty to very cherty limestone with a few thin sandy or shaly zones. For convenience Marcher and Kenny et al. (1984) collectively referred to these formations of the same age as the Mississippian aquifer.

(1) In most of its area of outcrop (NNE-SSW line east of Quapaw, OK, shown by Marcher and Kenny et al., Fig. 7.4-1) the Mississippian aquifer has been intensely weathered to a residual rubble of fractured chert. *Weathering and resultant slumping is important because the fractures thus formed provide openings for the entry, movement, and storage of water*. Localized areas of very intensely weathered rocks, referred to as breccia areas (Feder et al., 1969), have been mapped in some detail in the vicinity of Joplin,

Missouri. These breccia areas may yield as much as 300 gpm to individual wells whereas wells in nearby, less weathered rocks may yield only 5-10 gpm. *Breccia areas were the sites of mineralization by zinc and lead ores which have been mined leaving extensive underground openings now filled with water.*

(2) Because of vertical and lateral variations in hydraulic properties, water in the Mississippian aquifer occurs under both confined and unconfined conditions. Generally unconfined conditions occur in the upper 50-100 feet of the aquifer. However, there is little difference in water levels – commonly no more than 5 feet – in adjacent wells that are completed in confined and unconfined zones. In outcrop areas, the static water level commonly is less than 30 feet below the surface. *Where the aquifer is overlain by impermeable shale of Pennsylvanian age, the static level may be 50-100 feet below the surface depending on local hydrologic conditions.*

(3) Recharge to the Mississippian aquifer is derived mainly from *precipitation falling directly on the outcrop*. Average annual precipitation in Spring River basin, where the aquifer is widely exposed at the surface, is about 40 inches. According to Feder et al. (1969), about 5% of the precipitation recharges the aquifer, 20% runs off as streamflow, and 75% is lost to evaporation.

(4) Springs in the Ozark Plateau section of Area 40 are a very significant part of the hydrologic system. At least 50 springs discharging more than 100 gpm are known in the basin of Spring River (Vineyard and Feder, 1974) and many smaller springs and seeps undoubtedly are present. Springs discharge approximately 300 cubic feet per second (cfs, 194 million gallons per day, MGD) to Spring River and its tributaries thereby maintaining flow of these streams during times of no rainfall. Because of its uniform temperature (14-16º C), spring water is well suited for propagation of fish. Springs also supply water for domestic, stock, industrial, and recreation use.

(5) In the vicinity of Joplin, Missouri, abandoned mines are a source of water supply for various uses (Feder et al., 1969). Concentrations of dissolved solids in water from 23 mine shafts and open-pit ponds in the Joplin area ranged from 329 to 2,200 mg/l. Water from abandoned mines in Oklahoma and Kansas contains large concentrations of dissolved solids and toxic metals and, therefore, is unsuited for most uses without extensive treatment (Playton et al., 1980). The inability at the time (1982) for domestic water-treatment practices to remove toxic metals, such as cadmium and lead, precluded use of the mine water for public supply.

(6) Depending on local hydrologic conditions, excessively mineralized water from mines may move into the adjacent Mississippian aquifer. However, such movement apparently is not widespread in the Joplin area (Barks, 1977).

(7) Water quality from the aquifer typically is a calcium carbonate type, commonly hard to very hard. Water quality for wells and springs are provided in Table 8b.

Table 8b
Summary of Ground Water Quality in Mississippian Water-Bearing Formations from Wells and Springs, Area 40
(Modified from Feder et al., 1969 and Marcher and Kenny, et al., 1984, Tables 7.4-1)

Parameter	Wells				Springs			
	# Anal	Min	Median	Max	# Anal	Min	Median	Max
Bicarbonate	37	72	225	339	27	90	167	253
Calcium	39	33	80	221	29	31	55	147
Chloride	39	0.2	4.1	130	29	2.9	6.2	13
Dissolved solids	39	162	228	981	29	123	186	520
Fluoride	39	0.0	0.2	0.8	29	0.0	0.0	4.1
Iron (µg/l)	39	0.0	240	2,400	29	0.0	20	770
Magnesium	39	0.5	6.8	36	29	0.4	2.4	7.3
Manganese (µg/l)	39	0.0	0.0	200	27	0.0	0.0	100
pH (units)	39	6.1	7.9	8.3	27	7.0	7.5	8.3
Potassium	39	0.4	1.4	43	21	0.2	0.9	1.8
Sodium	39	3.0	7.6	106	21	1.1	4.3	10
SC (µmho/cm)	39	285	470	1,390	26	201	301	741
Sulfate	39	1.6	43	446	29	1.2	6.6	192
Zinc (µg/l)	38	50	900	6,700	--	--	--	--

All units in milligrams per liter (mg/l) 9.0unless noted.

c. Cambrian-Ordovician formation water yield and quality: This aquifer is an important source of water in the eastern part of Area 40. Yields of wells in the Cambrian-Ordovician aquifer typically range from 50 to 1,000 gpm and average about 200 gpm. Water quality is typically a calcium – magnesium bicarbonate type in the Spring River basin and Ottawa and Delaware Counties in Oklahoma. Farther west in Craig County, OK, the water changes to sodium chloride; and still farther west water is brine (very salty).

(1) Rocks of Cambrian and Ordovician age are present at greater depth throughout Area 40. However, these rocks are a water source (with a dissolved-solids concentration of 1,000 mg/l or less only in southwestern Missouri and adjacent parts of Kansas and Oklahoma. Farther west in Area 40, the Cambrian and Ordovician formations are the source of oil and gas and are used to dispose of oil-field brines and various industrial wastes.

(2) The Cambrian-Ordovician formations consist primarily of cherty dolomite with some sandstone, siltstone, and shale. These rocks were deposited on an irregular surface with local relief of as much as 800 feet (Ireland and Warren, 1946) so that units below the Roubidoux Formation or Gasconade Dolomite may be missing in some areas. Where all formations are present, their aggregate thickness is about 1,400 feet.

(3) Water in the Cambrian-Ordovician aquifer is confined. Wells in these formations generally are not cased below the top of the Cotter Dolomite so that all formations penetrated may yield some water. Consequently, the water level in a given well is a composite of all uncased formations. In Oklahoma and Kansas, the main water-yielding zones are sandstone or sandy dolomite in the Roubidoux Formation. In Missouri, the Eminence and Potosi Dolomites, as well as the Roubidoux, yield water to wells. The specific capacities of wells in the Cambrian-Ordovician aquifer in Spring River basin range from I to 20 gpm per foot of drawdown (Feder et al. 1969).

(4) The principal source of recharge to the Cambrian-Ordovician aquifer is in its outcrop area 10-30 miles east of Spring River basin. According to Harvey (1980), recharge is variable and occurs through sinkholes, by infiltration in permeable uplands, and by infiltration through streambeds. A secondary source of recharge is downward seepage from the overlying Mississippian aquifer.

(5) Water-level measurements in Spring River basin show that the potentiometric surface in the Cambrian-Ordovician aquifer generally is 50-100 feet below that in the Mississippian aquifer. Thus, if fractures or solution openings are present, water in the Mississippi – an aquifer will move downward. Such movement could result in the degradation of water in the Cambrian-Ordovician aquifer, particularly in areas where it is overlain by abandoned zinc and lead mines filled with excessively mineralized water (Fairchild and Christenson, 1982).

(6) The potentiometric surface in the Cambrian-Ordovician aquifer is highest in the eastern part of Spring River basin (see Marcher and Kenny et al., 1984, Fig. 7.5-1) near the areas of recharge. The slope of the surface and, consequently, the direction of water movement, is toward the west or northwest. Irregularities in the surface are the result of pumping. Pumping for municipal and industrial supply in the vicinity of Miami, Oklahoma, has produced a cone of depression with an area of about 500 square miles; a smaller cone has been developed in the vicinity of Baxter Springs, Kansas. During the early 1900's, the water level at Miami was near the land surface at an altitude of about 800 feet; a few wells reportedly flowed. However, long-continued and ever-increasing pumping has caused the water level to decline about 500 feet (see Marcher and Kenny et al., 1984, Fig. 7.5-2).

(7) Data for the chemical quality from the Cambrian-Ordovician aquifer in Spring River basin are summarized in Table 8c. These data also are representative of water from the aquifer in Ottawa and Craig Counties, Oklahoma. Water from some wells contains hydrogen sulfide, which gives it an unpleasant odor; however, the odor is readily eliminated by aeration.

The above information was the status of information as of 1984 and requires update of more recent information for background conditions of the wider basin in which the Tar Creek and Lower Spring River Watersheds are a smaller part. The need for monitoring of surface waters and ground waters (above, at and below) the Picher Field mining zone is most requisite. This is further discussed below in the section entitled: *Framework Monitoring Requirements (Networks).*

Table 8c
Summary of Ground Water Quality in Cambrian-Ordovician Aquifers, Area 40
(Modified from Marcher and Kenny, et al., 1984, Tables 7.5-2)

Parameter	# Anal	Min	Median	Max
Bicarbonate	38	120	206	257
Calcium	38	25	40	74
Chloride	38	1.7	5.2	22
Dissolved solids	37	140	227	290
Fluoride	25	0.1	0.1	0.5
Iron (µg/l)	37	0.0	700	1,700
Magnesium	38	11	18	22
Nitrate	36	0.0	0.0	12
pH (units)	35	7.3	7.6	8.4
Potassium	17	.6	1.5	2.0
Silica	37	5.0	8.0	12
Sodium	17	1.6	5.4	30
SC (µmho/cm)	5	347	369	474
Sulfate	37	3.7	13	68

Tar Creek Surface Waters – Flow & Quality

Data collection of discharge and water quality at various locations along Tar Creek and some of its tributaries has been ongoing but intermittent since 1979. The studies done by the OWRB (1989; ODEQ, 2004)) regarding surface flows via weir measurements, field water quality measurements, and mine water level measurements from the underground workings[7] shows observations that can be explained from a knowledge of mine water rebound (explained previously), and a better understanding of climatic influences on recharge to the mine working themselves.

Original OWRB investigations (OWRB 1982, 1983a,b) for EPA's Operable Unit 1 (OU1 – Surface Water/Ground Water) determined from the second five-year review (USEPA, 2000) that the original premise of recharge from Muncie Collapse and Big John Collapse as well as the area on Lytle Creek upstream of its confluence with Tar Creek contributed 75% of the surface flow to the underground mine workings was incorrect. The mechanism for recharge to the subsurface is through many more surface features such as abandoned shafts, drill holes and in particular collapse features.

Chat piles containing coarser grain-size fractions actually infiltrate more water than they shed through runoff processes. This infiltrating water will develop its own perched water table condition with larger chat piles and actually develop a flow system within it that can contribute shallow ground water as seepage at it base, deeper into the mine workings if located over collapsed shafts, drill holes or sinkholes, and as base flow to creeks and watercourses through continuity of the chat material lining these same channels or watercourses. Hence, what at first look may appear to be little surface water conveyance

[7] Represented by (1) monitoring the air shaft at the Blue Goose #1 mine, and (2) an open abandoned well casing at the Tar-Lytle Creeks' confluence (OWRB Site 4S) during the period 1981 to 1989.

to the subsurface is in actuality an entire system of conveyance to the subsurface and further water movement laterally or downward (should pathways exist via shafts, etc.) back to the mine workings.

In order to evaluate the influence of surface processes that primarily consist of climate effects of alternating wetting and drying periods on mine water levels, comparison of the mine levels to the Palmer Index[8] were performed for this period. Popularly known as the Palmer Drought Severity Index (PDSI), it is in actuality a very good indicator of both wet and dry periods and uses a surface mass balance approach to account for moist flux (or the lack thereof in drought conditions). The PDSI (as well as other forms of the index such as the hydrologic index, PHDI, and the modified index MPDSI) is computed monthly for each state's climatic division by the National Climatic Data Center (NCDC) of the National Oceanographic and Atmospheric Administration (NOAA). It can be found at the NCDC's Climate Visualization website (http://www.ncdc.noaa.gov/onlineprod/drought/xmgr.html) and applies to individual climatic divisions within each state. An example map of the Palmer index for the United States showing the climatic divisions for the Tri-State region is given as Fig. 9.

The Tar Creek and Lower Spring River Watersheds fall within three climatic divisions: Oklahoma Climatic Division 9, Kansas Climatic Division 3 and Missouri Climatic Division 4. For the present comparison, a "weighted" PDSI value based on proportioning the drainage area under consideration by corresponding state climatic division was used. Tar Creek Watershed (drainage area of 52 sq mi) headwaters originate in Kansas (16.84 sq mi drainage area or 32.4%) with the remainder of the creek in Oklahoma (35.16 sq mi or 67.6%). This proportionality by area makes up the "weighting factor" to apply to the corresponding PDSI values for Kansas and Oklahoma. Hence a weighted value of the PDSI was applied to the watershed for the period 1980 to 1993 and comparison made to the mine water levels. Trend data are shown in Figure 10a (separate figure at end of text). Palmer trend with the USGS gage record for the USGS gage in Miami (Central Ave bridge, 1980 to 1984) and subsequent change to the 22nd St Bridge location (1984-1993) is shown in Figure 10b (separate figure at end of text).

Figure 10a shows that there were a number of drought periods and wet periods between 1980 and 1989. As discussed previously regarding mine water chemistry, the adding and subtracting of moisture is key in driving the geochemical weathering of sulfide minerals as well as those (limestone and dolomite, chert host rocks) that buffer this acidity generation. The "drying out" periods represent slowing of the weathering process and deposition of the acid generating salts (AGS) that are then mobilized (flushed) from the system during "wet periods." This accounts for the better water quality readings during the period 1987 to 1989 and why it appeared quality was declining in severity (initial

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[8] The Palmer index (Palmer, 1965) was developed for semiarid (western Kansas, United States) to subhumid (Iowa) regions for assessing drought severity and is now applied over the whole of the United States. The analysis is based on a weekly or monthly water balance, and output from the analysis is an index value ranging from -4 (extreme drought) through 0 (normal condition) to +4 (extreme wet period). The index is computed as a function of the difference, accumulated through time, of the actual rainfall and the CAFEC rainfall (climatically appropriate for existing conditions of temperature, evaporation, and other components of the water balance). Evaporation is computed by the Thornthwaite method, and runoff, percolation, and soil-water levels, by a simple soil-water balance model.

dilution effect) but then suddenly got worse (flushing of the salts ensued) concentrating movement to the surface and recorded as declining water quality. The good news regarding mine water seepage is one can anticipate this condition, the bad news is it will continue until something is done to slow it down and reduce it, or stop it (which may be

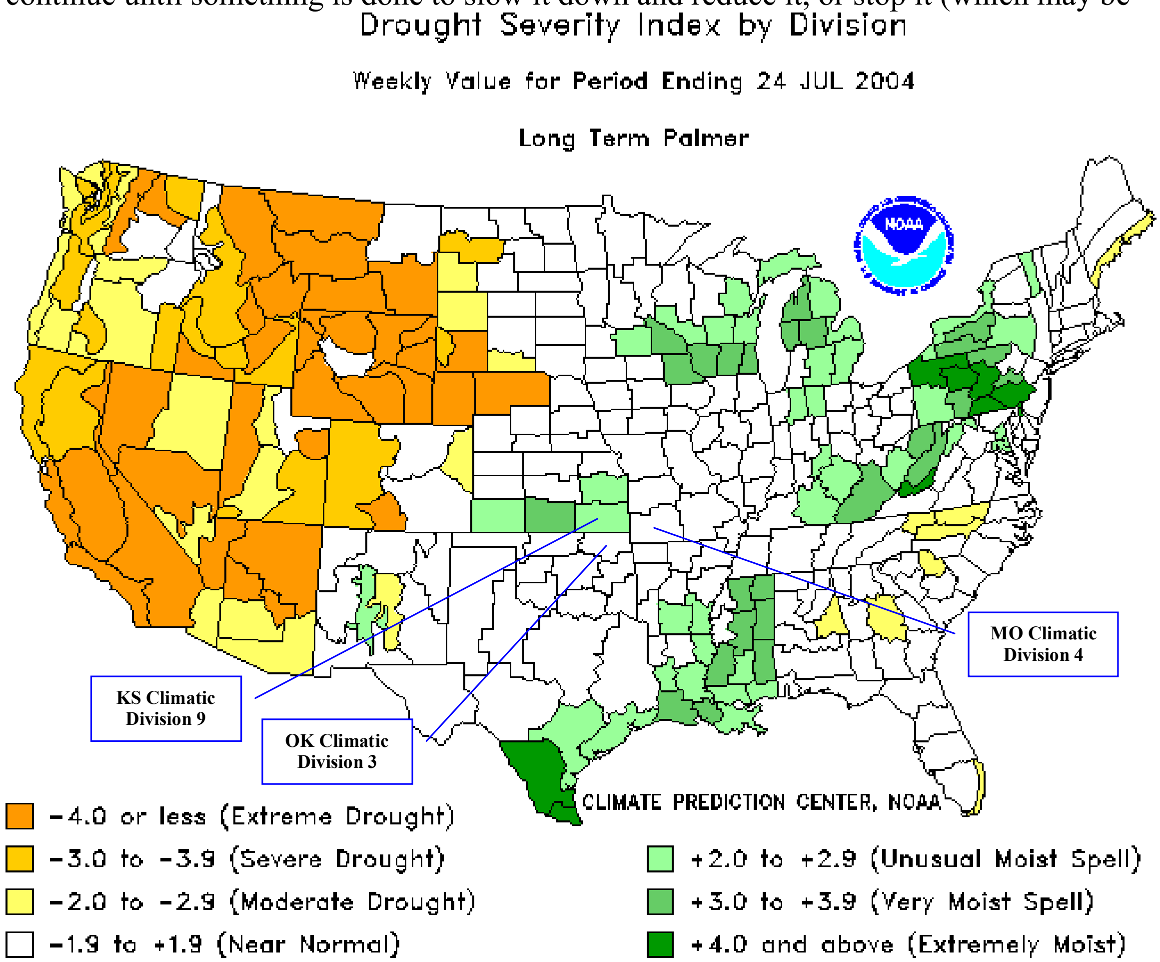

Figure 9. Example of the PDSI computed for the United States for each state's climatic division for the period ending 24 July 2004 (Image provided by NOAA). Relevant OK, KS and MO climatic divisions for the Tar Creek – Lower Spring River Watersheds shown in blue. MO and OK moisture conditions are normal whereas Kansas's moisture conditions for this period are above normal.

possible in some areas) and contemplate appropriate treatment of the mine-source surface seepage until the acid generating materials are depleted (which will likely take several decades if not longer). Improvement and preservation of surface water quality cannot be realized without this consideration for the Tar Creek Watershed.

Figure 10b was prepared to confirm the tracking of high and low flow with the Palmer Index. These data indicate a relationship of surface recharge to the underground mine workings is more sensitive to infiltration effects throughout the mine impacted area then first supposed in the early 1980s. A regression fit of the PDSI with mine water levels for the Blue Goose shows a linear fit explaining about 64% of the variation during the period 1980 to 1988.

There are numerous areas recharge can occur particular in close proximity to where all the seep features are found. In addition, areas far from the seep locations along Lytle Creek and into Kansas toward the northwest are frequented with numerous large collapse features with adjoining chat piles as well as open shafts conducive to surface infiltration. One misconception is that if a mineshaft is filled with chat or other coarse material it is "plugged." This may be true, but it is rarely sealed to water flow. In high surface flows along the creeks, it is reported high water overtops worn diking features and can run into abandoned open shafts. In addition, perched ground water infiltrated into large chat piles can prolong the drainage effect into the subsurface unless rejected by saturation at depth.

These are important occurrences that suggest flow conditions along both Lytle and Tar Creeks above the Hwy 69 bridge require proper designed channel ways to pass nominal flows (10-yr, 24 hr storm event as a minimum) to improve the surface conditions along their courses. As explained earlier, common practice in processing mine tailings was to wash them to the nearest watercourse. Historically this practiced caused poor flow conditions to be experienced by the streams to where backwater effects, clogged or slow flow conditions were prevalent and a change in practice to sedimentation ponds was called for. This has occurred both in Tar and Lytle Creeks above their confluence. This condition will be addressed by the Additional Activity entitled: Stream Corridor Restoration.

One other condition of recharge, which is unfavorable as long as chat and other tailing or mine wastes remain at the surface, is the quality of the surface water. Once infiltrated into these features begins, a degrading process can begin which generates AMD. Those mine wastes or tailings receiving the most processing prior to abandonment will likely be the "cleanest" from a residual sulfide availability perspective. For those that didn't, there remains the potential for degradation of infiltrating surface waters within them. Until these waters are buffered by dissolution of alkaline material (carbonates and silicates), their acidity will persist. Therefore, it is important to test the acid-generating potential of "chat" piles to determine their ability to generate AGS and be the source of AMD. Older mine workings, and those that did not receive multiple or improved processing are suspect for these conditions.

Figure 11a and b show the Hwy 69 bridge discharge (OWRB Site 10) and Blue Goose mine water elevation trend for the period 1980 to 1989, respectively. The drought period mentioned above particularly impacted mine water levels along Tar Creek numerous times stopping seepage from the mines (Site 4S) which are shown on both figures. However, it did not stop flow at OWRB site 14S (Mayer Ranch). The drought events with relevant durations are shown in Table 9.

Figure 11b shows the elevation point (end of callout line) flow stopped in relation to the minimum mine water level experienced during the corresponding no flow period. Table 9 lists the elevation condition that corresponds to when flow started again at OWRB site 4S. As can be seen in Figure 11a, wet conditions prevailed most of 1985, beginning and ending in 1986 and in the beginning of 1987, and again at the end 1987 and beginning 1988. This record can be analyzed past 1988 as the data is available, but time constraints in processing precluding evaluation of more of the record. This needs to be done prior to any seepage control design activities.

Table 9 PDSI Drought Periods – 1980 to 1988, Tar Creek area, Picher Field, OK						
#	BG Mine W.L. Elev. Flow stopped	Start Date	End Date	BG Mine W.L. Elev. Flow started	# Drought Days	Max PDSI (Date)
1	796.97	5 Jul 1980	13 May 1981	797.58	312	-4.34 (4/30/81)
2	796.82	6 Oct 1981	13 Oct 1981	796.75	7	-2.68 (9/30/81)
3	796.93	7 Sep 1982	7 Dec 1982	799.07	~91*	-1.45 (10/31/82)
4	796.81**	25 Aug 1983	20 Oct 1983	799.23	56	-2.12 (9/30/83)
5	796.91	2 Aug 1984	31 Oct 1984	796.62 **	90	-2.01 (9/30/84)
6	798.31	9 Sep 1988	14 Sep 1988	798.39	5	-2.17 (8/31/88)

* 26 Nov to 5 Dec no data, likely flow commenced 7 Dec 1982.
** Minimum flow recorded to stop and start flow at OWRB Site 4S (just east of Tar-Lytle confluence).
Mine W..L. expressed as ft above mean sea level.

Table 10 (separate table at end of text) summaries the flow and field water quality for the 1980 to 1989 period. As can be seen for individual mean values, water quality generally improved during the 1984 to 1989 period for sites above the Hwy 69 bridge and declined for sites below this point except at the confluence with the Neosho River which was probably due to high flow backwater effects influencing these measurements. However, for mine seepage dissolved oxygen, concentrations reverted to pre-1984 levels indicating a flushing was occurring in the mine cavities and mixing of stratified mine water was occurring. This phenomenon is discussed in the *Mine Water within the Boone* section below. Better measurement of these effects over time at several fixed locations during the course of remediation activities is paramount (se section of *Framework Monitoring Requirements* below).

Tar Creek Ground Water – Flow & Quality

In considering ground waters, three perspectives are needed: (1) what is the impact above the flooded mine workings, (2) what is the conditions below the flooded mine workings; and (3) what is the condition within the mine workings regarding flow within void zones (mined out areas or mine workings) and laterally within the Boone formation where mining didn't occur. General area water quality for the geologic formations was previously presented after Marcher and Kenny *et al.* (1984). These typical chemistries persist in the Tar Creek area.

Extensive efforts have been expended to monitor and protect wells to the underlying Roubidoux Aquifer where most of the water supply from ground waters is supplied. These efforts appear adequate for the moment and no further comment is made here.

However, concerns for lateral flow (to and from the underground mine workings "void zone" which is currently water filled) and upwards-vertical flow (as most certainly is happening where known seeps occur but less so where they cannot be seen) have received relatively little attention. The lateral and upward pathways may be both a help and hindrance from a mine-water flow control and chemistry perspective. The objective of any subsurface flow control is to reduce the amount of poor quality water evading the shallow subsurface and showing itself in the surface flow environment primarily because of its poor quality and degrading effects on surface (oxygen-rich) environs.

At least three shallow subsurface conditions requiring determination, relative to the Boone, exist (from east to west starting at Spring River moving west): (1) one involving where recharge effects the Boone formation directly on its outcrop areas (generally on a line NNE-SSW, 2-miles east of Quapaw, OK eastward which stops just west of the Spring River trace), (2) a transition area where the overlying Batesville Sandstone and Hindsville Limestone occurs (edge of the Upper Mississippian rocks terminating near Quapaw, and lying between the Boone outcrop to the east and the younger Krebs-Hindsville Formations to the west), and (3) that area under the Krebs-Hindsville Formations. Particular for the Tar Creek-Lytle Creek confluence, the second condition above occurs where a remnant outcrop of Batesville sandstone and Hindsville limestone or Quapaw limestone occur just east of the confluence. These formations contribute to the conveyance and buffering effects that can counter AMD effects.

Current efforts by the U.S. Geological Survey are focused on developing a ground water model of the mine workings and these overlying (as well as the lower Roubidoux) conditions. In this regard, relational geologic and hydrologic design data are needed to explain relations of this complex flow system. Establishment of well locations and monitoring of these conditions is most needful for the modeling effort to be successful in complementing evaluation of the water control alternatives. Suggested monitoring of the groundwater regime, other than the Roubidoux is provided in the *Framework Monitoring Requirements (Networks)* section below

Mine Water within Boone – Flow & Quality

Understanding what happens when mine water floods abandoned underground workings is necessary to relating the type of water quality expected over time at seepage points. Prudent remedial design requires this be known in order for designs and behavior of the correction action have certainty over time.

When mine dewatering ceases, the water make of the mine does not. Hence the mine voids gradually fill with water, in a flooding process which has come to be termed "rebound" (as described previously) Rebound will continue until such time as the water levels in the mine voids arrive at one or more "decant or surface or near-surface seepage points". As the mine water decants (i.e. overflows), into an adjoining aquifer, formation and/or into a surface water body, water levels in the mine voids stabilize, and will thereafter tend to remain within a narrow range (responding to seasonal fluctuations in water infiltration or “make”). The mine workings at Tar Creek are currently in this state. Predicting the details of rebound (and corresponding seepage behavior) in a given situation is a complex task. Before it can even be considered, it is necessary to develop a conceptual understanding of processes of water movement and the associated hydrochemical changes during the process of rebound. First, it is important to appreciate that there are critical differences between the mode of hydrogeology encountered in working mines (dewatering and keeping water out) and the hydrogeological behavior of mined systems during rebound (flooding once keeping the mine dry has ended). These differences are largely attributable to one fact: in working mines, the huge void space comprising the mine itself are kept dry by strategic pumping, whereas during rebound, the mine voids themselves become the principal conduits for water movement.

Predict mine water rebound must account for flows within the mined voids, as well as flows in the strata beyond the voids (lateral flow to and from host rock supporting the mine area). During rebound, when hydraulic head gradients are usually high, flow through the large, open mine voids, can be confidently predicted to be turbulent. As mentioned previously, for this condition, it is necessary to use the analysis of the hydrogeological behavior of karst conduits developed by Smith *et al.* (1976). It is clear that flow in most open mine voids is likely to be turbulent whenever flow velocities exceed about one millimeter per second (mm/s = 0.003 ft/sec). This has at least two important consequences:

1) Turbulence favors erosion of the mine voids. This can have negative consequences, such as the collapse of previously stable horizontal levels (called roadways). On the other hand, erosion by rapidly flowing mine waters above the water table has been observed to prevent the blockage of open roadways which would otherwise have been sealed by floor heave processes (Younger *et al.*, 2002). Erosion typically results in the mine waters carrying a high suspended-solids load. An example of this phenomenon is reported from Scotland by Younger and LaPierre (2000), in which rebound in one colliery led to distinct peaks in suspended solids in the water drained from two adjoining mines, as the water seeped into each of them in turn from the rebounding mine.
2) The occurrence of turbulent flow violates the assumption of laminar flow upon which all major groundwater flow modeling packages are based (e.g. Anderson and Woessner, 1992). Turbulent flows cannot be simulated realistically using such packages, necessitating the use of alternative model formulations.

- *The Concept of Ponds*

During mining, the contrast in permeability between intact strata and near-void strata affected by subsidence is a matter of major interest. The contrast in this case is generally about three to four orders of magnitude in terms of hydraulic conductivity. The contrast between near-void strata and the voids themselves spans no fewer orders of magnitude, albeit open voids are so "permeable" they fall out with the bounds of Darcian classification (Younger and Adams, 1999). If we go on to contrast intact strata with open mine voids, the contrast in terms of hydraulic conductivity is in excess of seven orders of magnitude.

Recognition of this marked contrast is nothing new. Indeed, it has long been implicit in the actions of surveyors and engineers in the coal industry, particularly in the manner in which they conceptualize volumes of interconnected workings as 'ponds' (or mine pools of water) separated by barriers of unworked coal (or in the case here ore; Minett, 1987). Inherent in the definition of a pond is the concept that the mine-workings within anyone pond are extensively inter-connected (often on multiple levels) so that water rising within anyone pond will display a common level throughout that pond. At certain elevations, adjoining ponds may be connected by way of discrete decant (or spilling) features. Typical decant features include:

- roadways inter-connecting areas of otherwise discrete workings,
- areas in which two adjoining floor or roof-hanging rock panels coalesce,
- old exploration boreholes,
- permeable geological features (e.g. the margins of a limestone bed, brecciated chert zone, or an open fault plane).

Systems of ponds can be defined at a variety of resolutions. This leads to the notion of looking to see where parts of the mining area are connected or not connected. For example, the mine workings in south Commerce are separated from those in the main field to the northwest. Rebound in the south Commerce field was different than for the main Picher Field. Outlying mining areas not connected to the main Picher Field will have different rebound characteristics than those in the heavily connected, main mining area.

Implicit in the recognition of ponds is an expectation that the process of rebound will occur independently in two (or more) adjoining ponds until such time as the water level in one or more of the ponds reaches an inter-connecting decant feature (see Figure 12). Inter-pond transfers of water will then occur until the difference in water level (head) between the two ponds either side of each decant point is minimized. If the decant is "unrestricted" (i.e. offers very little resistance to inter-pond flow, e.g. a large-diameter roadway), then the ultimate head difference between the adjoining ponds will tend to zero. Where the decant is "throttled" (i.e. resists flow to some degree, which might be the case if the feature in question is a hanging body of rock, a narrow borehole, or a natural geological feature), the ultimate head difference between the two ponds may amount to a difference of several feet or ten's of feet. Differentiation between unrestricted and throttled decants is possible where a record of seasonal water levels in the pond with higher water level is available. A pond with an unrestricted decant will transfer newly-recharged water to the adjoining pond so rapidly that water level fluctuations will be minimal (less than a few feet), whereas seasonal water level fluctuations in a pond with a throttled overflow may be quite marked (3 to 70 ft; Younger *et al.*, 2002).

- *Controls on the rate of rebound*

The rate of water level rise in a given pond is a function of two factors:

1) The distribution of available water storage volume within the pond and
2) The rate of water inflow to the pond,

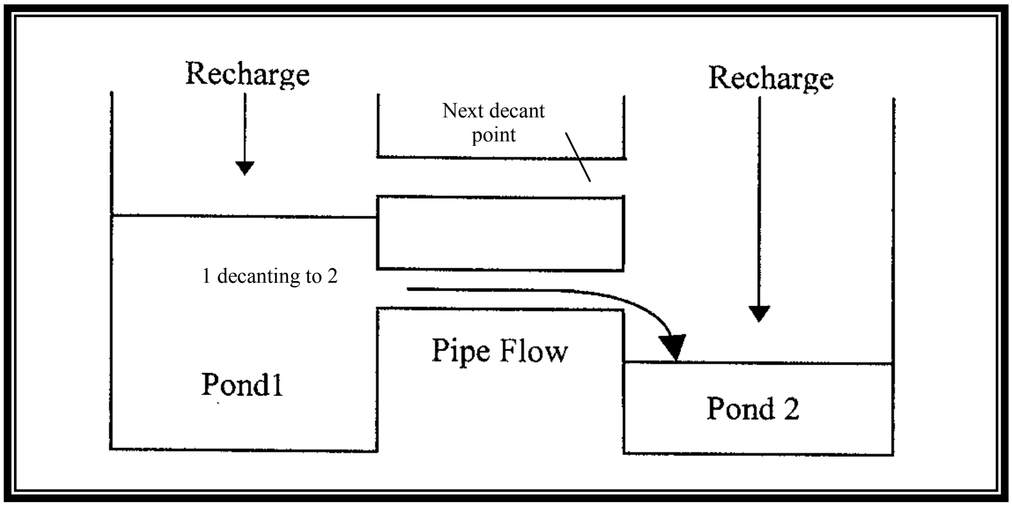

Figure 12. Concept diagram of mine working's cross-section illustrating the concept of mine water rebound in two adjoining "ponds" or working areas which are connected only by two discrete decant routes (e.g. roadways, or areas where hanging panels coalesce). Until the water level in at least one of the ponds reaches the lowermost decant route (pond 1 in diagram is shown passing that point, the rebound process will occur independently in each of the ponds. Thereafter, water will flow from one to the other (as shown) eventually resulting in an equalization of heads between the two, and subsequent common evolution of rebound in both ponds. (Modified from Younger et al., 2002 and Sherwood, 1997; see also Sherwood and Younger, 1997).

In most cases, the available water storage volume will equate to the sum of the volume of mine voids and the porosity of the enclosing rock mass (provided the latter was actually drained during mining). For surface mines, the volume of mine void space is easy to define and the deeper the mine, the more complicated the relationships.

The overall rate of water inflow to a given pond can be thought of as the sum of two categories of inflow:

1) head-dependent inflows, and
2) head-independent inflows

Head-dependent inflows account for most of the water make in the majority of sub-water table mines. They represent inflows from adjoining aquifers, be these above, beside or below the mined voids (Younger *et al*., 2002). The rates of inflow from these aquifers depend on the degree to which the hydraulic head (water level condition in formation) within them exceeds the head (elevation plus atmospheric pressure) in the mine voids. The head difference between a given aquifer and an adjoining mine void will clearly be at a maximum when the mine is fully dewatered to its entire depth. As rebound progresses, the head differences between the aquifers and the mine voids will progressively decrease, leading to both a gradual reduction in the rate of inflow and a gradual deceleration of water level rise. Where head-dependent inflows predominate, these processes will result in the rebound curve being exponential in form (case (1) in Figure 13).

Head-independent inflows enter mine voids from remote sources via long recharge

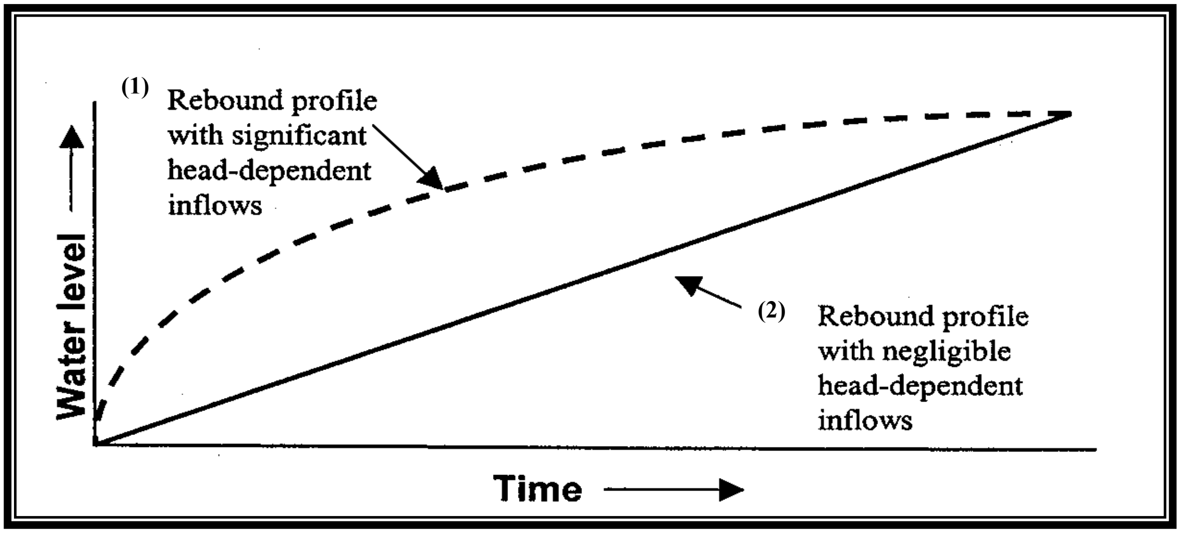

Figure 13. Schematic mine water rebound curves which would be anticipated for the two extreme possibilities of (1) a mine system which receives a large proportion of its total inflow from head-dependent sources (i.e. inflow from adjoining aquifers which initially have a higher head than the mine system), and (ii) a mine system in which virtually all of the inflow originates as infiltration from the land surface above the mined area, or by decant from an aquifer which is perched above the mined system and drains into it at a constant rate. (Modified from Younger *et al.*, 2002 and adapted from Sherwood, 1997).

pathways. This means that the rate of inflow to the mine voids is independent of the water level within them, depending only on the availability of water in the remote source. The principal sources of head-independent inflows to mine voids are:

- Rainfall directly into the void (surface mines or underground mines open to the surface at many locations) or infiltrating into strata immediately overlying the void (underground mines)
- Surface water courses which cascade into the void (surface mines or openings to underground mines) or infiltrate into strata above the void (underground mines)
- Drainage from aquifers that are perched high above the worked strata, such that the aquifer base lies far above the highest level which water will ever reach in the mine void.

Where head-independent inflows predominate in the water make of a rebounding mine system, the rebound is likely to be approximately linear (case (2) in Figure 13).

From the results of Playton *et al.* (1980, see Fig. 13 for the Blue Goose rebound curve, p. 46), recharge to the mine workings in the Picher Field is head-independent inflows which further verifies the results of the Palmer comparative analysis discussed previously.

Hydrochemical changes accompanying rebound consist of (1) water quality deterioration and (2) stratification of the water column within the mine workings.

It has already been noted previously (see *Mine Water Chemistry* section) that rebound commonly results in a marked deterioration in the quality of mine waters. This is due to the sudden dissolution of the various minerals (see Table 4, secondary mineralogy) that comprise the "vestigial acidity" in the workings. As a general guide, this problem is only likely to give rise to significant contamination where the mined strata contain greater than 1% by weight total sulfur. In such cases, dissolved concentrations of sulfate, iron and/or other problematic metals can be expected to increase by as much as an order of magnitude during rebound (Younger, 2000a).

This deterioration in water quality has important environmental consequences when the abandoned workings finally overspill to the surface environment as has already been experienced within Tar Creek.

There is an important nuance in the hydrochemical changes that accompany rebound, namely, ***stratification of water quality***. This was apparent from the work of the USGS when rebound was occurring (1975-76, Playton *et al.*, 1980) and again recently (DeHay, 2002). Although water quality in the mineshafts was found to be somewhat improved, stratification nevertheless persisted.

It is immediately apparent in both USGS mineshaft-sampling studies that the better quality water is found at the top of the water column, albeit step changes in the different water quality parameters do not always coincide. Other examples of stratification are presented by Ladwig *et al.* (1984), from anthracite mines in Pennsylvania, and Johnson and Younger (2000) from a fluorspar mine in northern England.

The stratification of water quality during rebound implies that mechanical mixing of the water column in the mine is minimal. This in turn implies, for instance, that there are few lateral inflows and outflows at depth. As long as flow remains sluggish and laminar, there is little to disturb the stratification. However, when discharge from a mined system commences, turbulent flow in the vicinity of shafts and open roadways can cause substantial mixing of the mine water body, leading to a breakdown in the stratification. *Hence the quality of a mine water discharge from a formerly stratified system is more likely to resemble a mixture of all depth intervals rather than the better quality water previously found at the top of the water column.*

This has two important and practical implications:

1) Mine water discharges may well be considerably poorer in quality than would have been inferred by sampling the uppermost waters alone. Hence depth-sampling is highly advisable in studies of mine water rebound.

2) Where stratification is identified, it would be imprudent to assume that future discharges at the surface will resemble the uppermost waters in the rebounding water column. While the very first waters to emerge may have this benign quality, these will usually be quickly followed by waters of a more mixed nature, with an overall quality approaching that of the mean concentrations found over the full depth of the mine water body.

- *Longevity of Pollution from Abandoned Picher Mines*

After an abandoned mine decants to the ground surface or an adjoining aquifer, the quality of the water flowing from the workings usually improves over time, until some long-term 'asymptotic' level of residual contamination is reached (Figure 14). The initial improvement in quality, which may take years or even as long as four decades to reach completion (Wood *et al.*., 1998; Demchak *et al*., 2000), has come to be termed *"the first flush"*' (Younger, 1997a). The word "flush" is appropriate, since the water quality improvement corresponds to the gradual flushing of the voids, such that the highly

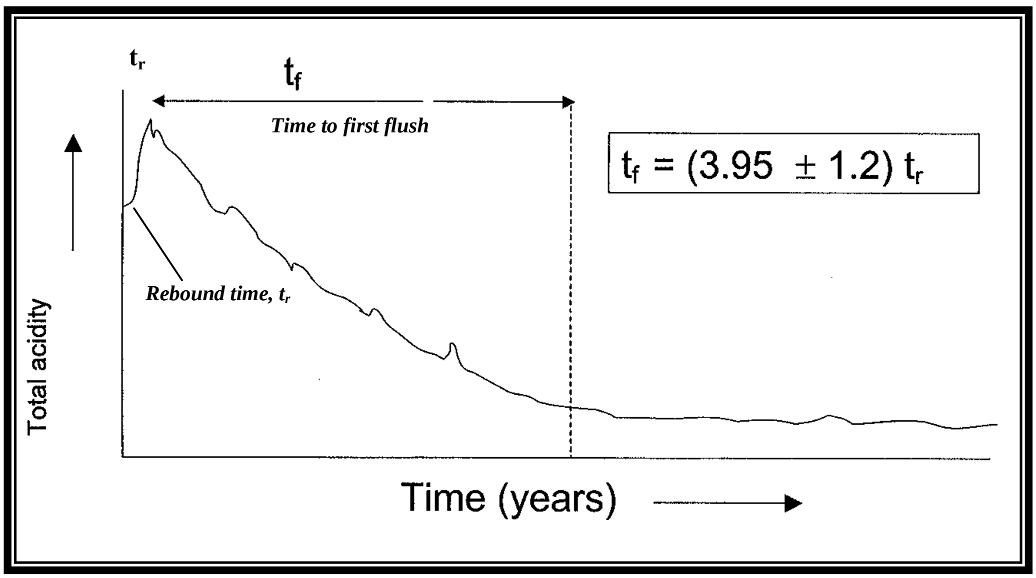

Figure 14. Relational graph illustrating the '*first flush*' phenomenon and its duration. t_f is the duration of the first flush and t_r is the '*rebound time*', i.e. the time it took the workings to flood up to surface decant (seepage condition). (After Younger, 2000a; and Younger, 1997a)

contaminated waters which typically occupy the voids immediately after the completion of rebound are progressively displaced by fresh, less contaminated recharge. After completion of the first flush, the long-term "'asymptotic" quality is determined by the rate of ongoing pollutant release in and near the zone of water table fluctuation (Younger, 1997a, 1998b).

The balance of factors that determine the nature (and temporal changes in) the long-term asymptotic water quality are the relative weathering rates of pyrite (marcasite or other sulfides) and calcite (dolomite and chert) as well as the critical parameters of the times to depletion for pyrite and calcite. Approaches to determine these are given by Younger et al. (2002, pp. 104-112). These weathering and mass mineral analyses should be developed for the Picher Field in connection with any mine water treatment system design.

Considering the predominantly hydrological factors that influence the form and duration of the first flush, Glover (1983) seems to have been the first to appreciate that the first flush is readily explicable in terms of hydraulic flushing, when he drew upon extensive personal experience of mine waters in the UK to suggest that the iron content of an abandoned mine discharge will typically halve in each subsequent period of time equal to that taken for the abandoned workings to fill with water after the pumps were withdrawn (i.e. an exponential decline). As discussed previously regarding underground "pond" filling behavior, the time mine workings take to flood is a function of the rate of water inflow and the volume of voids. These are precisely the same hydrological factors which will broadly govern the rate at which flooded mine workings can be flushed by fresh recharge (Younger, 1997a). Scrutiny of more recently available records largely vindicates the suggestion of Glover (1983), such that Younger (2000a) has proposed that:

the first flush is generally exponential in form, and

the duration of the first flush (t_f can be related to the duration of the rebound process (t_r as follows (Figure 14):

$$t_f = 4\ t_r \tag{12}$$

Hence, using the time to recovery of Playton *et al.* (1980, Fig. 13, p. 46), the mine area around the Blue Goose mineshaft recovered in approximately 53 months (Sep 1975 to Feb 1980). This is the value of t_r for this area. This suggests a time to first flush of almost 18 years ± 5 yrs. This agrees well with the estimate of Parkhurst (1986, p. D-6) of 22 years as the residence time for waters in the mines based on an annual volume of mine discharge of 3,400 ac-ft/yr for a mine volume of 76,000 ac-ft (OWRB, 1983a). Longevity of contamination levels will depend on weather rates and depletion of pyrite (and other sulfides) and calcite (dolomite and chert) within the mine workings.

Framework Monitoring Requirements (Networks)

Surface and ground water monitoring of the Tar Creek area (including areas in Lower Spring River where the Boone outcrop occurs) should be planned in order to protect the hydrologic balance of the area before, through and note the changes after the various remedial (On-going and Additional) Activities are performed. A two tier water monitoring approached should be considered, a wide area monitoring array (framework system) of fixed surface water stations and ground water well locations should be maintained by the U.S. Geological Survey, and local monitoring attendant to the remedial actions under consideration and responsible for by the sponsor(s) of the local project work. The local monitoring systems should tie in to the framework system where feasible such that interpretation of hydrologic effects can be assessed after remedial actions are completed. Adaptive management strategies can then be utilized to provide further corrective actions, if necessary.

Suggested Statement Hydrologic Balance Protection

Remedial and reclamation activities should be conducted so as to:

1) Minimize disturbance of the hydrologic balance within the remedial area proper and adjacent areas unless the purpose of the remediation is to improve hydrologic function of the area under consideration.
2) Prevent material damage to the hydrologic balance outside the remedial area proper.
3) Support proposed, approved or acceptable existing land uses in accordance with the terms and conditions of the planned actions of the agency (or agencies, public, private of non-profit) performing the remedial action.
4) Additional preventative, remedial or monitoring measures may be required to assure that material damage to the hydrologic balance outside the immediate area under remedial consideration. Those reclamation practices that minimize water pollution and changes in flow (unless necessary to improve hydrologic flow, prevent flooding or other damages) are preferred to water treatment.

Suggested Statement for Ground Water Protection

1) GW quality should be protected by handling earth materials and runoff in a manner that minimizes acidic, toxic or other harmful infiltration to ground water systems and by managing excavations and other disturbances to prevent or control discharge of pollutants into ground water.
2) GW Monitoring: A monitoring plan should be planned and followed by each project sponsor and should take advantage of existing monitoring devices or systems and anticipate planned adjacent projects so as not to compromise ongoing or recently completed remedial projects. Monthly or quarterly frequencies should be considered appropriately in relationship to the duration of the project under consideration. Minimum parameters to be considered for monitoring should consist of basic cation and anion chemistry, acidity and alkalinity, total and dissolved iron and manganese, and the field constituents: pH, DO, specific conductivity and temperature. Other parameters can be added on a case by case basis depending on the specific actions being taken.
3) Monitoring activities can cease if the purposes of the monitoring plan have been fulfilled for the intended actions being performed.

Suggested Statement for Surface Water Protection

1) To protect the hydrologic balance, remedial activities should be conducted according to the remedial plan of the sponsoring agency (public, private or non-profit).
2) SW quality shall be protected by handling earth materials, ground water discharges, and runoff in a manner that minimizes the formation of acidic or toxic drainage, prevents to the extent possible using best technology currently available, additional contribution of suspended solids to stream flow outside the remedial action area, and otherwise prevent water pollution.

3) Drainage from acid- and toxic-forming materials and underground development waste into surface water and ground water shall be avoided by:
   a. Identifying and properly disposing and/or treating, when necessary, materials which may adversely affect water quality, or be detrimental to vegetation or to public health and safety if not properly disposed and/or treated; and
   b. Storing materials in a manner that will protect surface water and ground water by preventing erosion, the formation of polluted runoff, and the infiltration of polluted water. Storage shall be limited to the period until proper disposal (planned disposal) and/or treatment first become feasible, and so long as storage will not result in any risk of water pollution or other environmental damage.
4) Surface water quantity and flow rates shall be protected or modified to be protective by handling earth materials and runoff in accordance with planned actions protective of on-site and adjacent surface water resources.
5) Surface Water Monitoring: A monitoring plan should be prepared to sample water quality monthly for planned actions less than 1-yr and quarterly (seasonal, every three months, as a minimum) for actions greater than 1-year to provide feedback for improving or correcting actions being taken or having adverse impacts to the hydrologic balance (unless specifically intended to improve the balance). NPDES criteria shall be used where applicable and governed by regulation for the practice being performed. Monitoring shall be maintained before, during and for the prescribed period after remedial actions to verify reclamation performance. Parameters for selection and frequency of monitoring shall be commensurate with ground water requirements and if none are necessary for ground water protection shall consist of the ground water parameters plus total suspended solids and flow discharge.

Other criteria for hydrologic protection and proper design for remedial construction activities shall be determined prior to remedial actions.

The framework monitoring is recommended to be conducted by the USGS and should consist (as a minimum) of permanent surface water stations (discharge & quality) on Tar Creek at the following locations: (1) near the Oklahoma-Kansas state line; (2) at the Hwy 69 bridge (currently being installed), (3) at the 22nd St bridge and (4) at the Hwy 10 bridge. Temporary gaging at the Tar-Lytle diversion and spring flow at the Douthat bridge. For the Lower Spring River Watershed: (1) Maintain existing station above Beaver Creek confluence, (2) at the spring in the upper basin of Beaver Creek (currently being installed), and (3) near the KS-OK state line.

Ground water monitoring wells should be planned for the Boone Formation surrounding the mining area with at least three existing and/or new monitoring wells per cardinal direction (N, E, S, and W) around the former mining area to develop flow conditions (potentiometry, nature of flow as fracture or porous, storage properties) in areas adjacent to the underground workings. At least three wells should be placed in the Boone adjacent to the west side of the Spring River to ascertain recharge properties and vertical flow relationships in the recharge area. At least three additional wells should be planned for the Batesville Sandstone and Hindsville Limestone area to ascertain lateral and vertical relationships to the Boone Formation. Geophysical techniques could be considered to offset costs to ascertain geologic relationships to avoid drilling.

Roubidoux monitoring is adequate.

At least two other monitoring points should be set up in the mine workings themselves besides the Blue Goose station. These monitoring locations should be near the southeast and northern portions of the Picher Field. Precise locations should be determined after consideration of Boone monitoring well placement.

Other framework monitoring should be considered once On-going Activities are better defined and Additional Activities alternatives are decided.

Tar Creek – Lytle Creek Recon Remediation

General Remediation Techniques

Hydrologic interventions to prevent or minimize aqueous pollutant release from mine sites may be broadly classed into two categories:

1) Passive prevention of pollutant release, and
2) Active mine water control

Control & Treatment Strategies

Passive prevention can be formally defined as (see PIRAMID project at www.piramid.org)

"Passive prevention of pollutant release from mine sites is achieved by the surface or subsurface installation of physical barriers (requiring little or no long-term maintenance) which inhibit pollution-generating chemical reactions (for instance, by permanently altering redox and/or moisture dynamics), and/or directly prevent the migration of existing polluted waters".

Specific techniques within this category are many and varied, though most fall into one of following four subcategories:

- Submergence techniques
- So-called "dry covers"
- Subsurface impermeabilization, and
- Water diversion by gravity drainage or pumps.

A brief explanation of each of these follows.

Submergence techniques are possibly the simplest of all hydrological interventions for the remediation of polluted mine waters. This technique is to accept (or deliberately arrange) the flooding of mine voids in pyritic strata. This serves to cut off the supply of oxygen to submerged pyritic zones, thus eliminating pyrite oxidation in those zones. In addition considerable research has gone into the design of water tight seals for mine roadways (horizontal or inclined levels) to facilitate flooding of zones that would otherwise be free-draining (Fernandez-Rubio *et al.*, 1987; Ackman, 1987; Fuenkajorn and Daemen, 1996).

Some of the most successful applications of submerged techniques have been in remediating waste rock piles and abandoned tailings dams, in which context they are known as "water covers" or "wet covers."

The use of water covers to inhibit the leaching of acidity from waste rock/tailings is predicated on the fact that the rates of oxygen diffusion in water are dramatically lower than in .the atmosphere itself. The design of water covers is deceptively simple: all that is required is that some form of impoundment is arranged to ensure that a minimum depth of water is maintained above the surface of the flooded waste materials at all times. The minimum depth recommended is one meter (Younger et al., 2002), for two reasons:

1) where water covers are shallower than one meter they are usually so well-mixed that the oxygen content at the sediment surface is little less than at the water surface (Li *et al.*, 2000); in fact, there are reasons to advocate water depths of several meters if this is consistent with other site constraints (access, safety, etc).
2) where water covers are shallower than one meter, the surface of the underlying tailings is prone to agitation by wind-blown waves of the magnitude that can develop in impoundments with areas up to several hundred hectares[9] (i.e. of sizes appropriate to mine waste management situations). Re-suspension of tailings can result, leading to complications in water quality management (Catalan et al., 2000).

While minimizing the downward diffusion of oxygen is the principal objective of water cover design, upward diffusion of metal contaminants is also an issue, particularly where previously dry tailings have to be newly flooded at the time of water cover establishment. In such cases, the process of flooding causes the dissolution of acid-generating secondary minerals (i.e. the 'vestigial acidity' as discussed above) comprising minerals such as those listed in Table 4, secondary minerals). Gradual diffusion of these metals into the overlying standing water can give rise to problems in complying with environmental regulations.

One possible mitigation method for this circumstance has been suggested by St-Arnaud (1994), who showed that addition of a layer of permeable quartz sand to the surface of the flooded tailings can help to minimize the release by diffusion of metals to the water column. The process responsible is perceived to operate as follows: dissolved ferrous iron

[9] 1 hectare = 2.47 acres = 107,640 square feet.

tends to oxidize briskly once in the aerated open waters, forming insoluble ferric hydroxide (ochre). The structural framework provided by the quartz sand acts as a locus for accumulation of the ochre. As ochre has a very high sorptive capacity for most other metals (Dzombak and Morel, 1990), this sand-ochre layer can trap great amounts of contaminants that would otherwise be released to the open water.

The problem of metal diffusion from recently flooded tailings is one example of a wider drawback of submergence techniques in general, which is even more problematic where they are applied to deep mine voids. In other words, while flooding will prevent future oxidation of pyrite, it serves only to dissolve the existing products of former pyrite oxidation. Hence it can confidently be expected that the full water quality benefits of mine flooding (Fernandez-Rubio *et al.*, 1987) will not be realized until the first flush is complete (Younger, 2000a).

Two other problems apply only to the use of submergence techniques for mine voids (and especially deep mine voids) as opposed to waste rock piles or tailings dams..

1) Since total submergence can often be effectively impossible to arrange in the case of large systems of deep voids, there is a tendency post-flooding for the water table to fluctuate some distance below ground. If the zone of fluctuation is in pyritic rocks, and is shallow enough to be in close connection with atmospheric oxygen, the seasonal rise and fall of the water table can lead to an ongoing release of' "juvenile acidity" (see *Mine Water Chemistry* section and Fig. 6).
2) Retention of ground water at high pressures behind subsurface impoundments can be very difficult to ensure, both for reasons of the integrity of mine roadway plugs etc, but also because, as long as recharge continues, ground water head will continue to build up behind the impoundment until it overtops and decants via a natural outcrop or some other mining feature.

Dry cover are barriers to moisture and oxygen ingress (entry). As with water covers, the use of dry soil covers above reactive waste rock/tailings seems deceptively simple upon first inspection. At their simplest, dry covers can be used to achieve "dry entombment" of a body of mine waste by means of covering it with an impermeable cap: if no water can contact the waste, no leachate can be generated. In this type of application, dry covers do not differ at all from conventional landfill caps.

Subsurface impermeabilization approaches consists of preventing acid leachate generation within mine wastes by destroying their permeability: If moisture cannot access the acid-generating parts of the waste, then acid cannot be generated. Implementation of this approach requires at least four steps (Younger *et al.*, 2002):

1) Assay the body of waste for pyritic sulphur content, by means of closely-spaced boreholes and lab analysis of cuttings
2) Identify the zones with a pyritic sulphur content over some critical threshold (we suggest: > 0.5 weight % pyritic sulphur, or > 1 weight % total sulphur), possibly using geostatistical interpolation methods

3) Inject a grout under pressure into the most pyritic zones (taking care to use a sulfate-resistant grout), and
4) Monitor the leachate afterwards to ensure that acid generation is minimized; otherwise re-evaluate/repeat steps (1) through (3).

Despite the simplicity of this approach, it has apparently not been implemented (or its implementation has not been reported) at large scale. However, it is a technique with considerable promise for dealing with leachate release from spoil heaps which have previously been successfully afforested or otherwise revegetated, such that the installation of a new dry or water cover is unlikely to be a popular option.

Control of polluted mine drainage waters by gravity drainage is also known as "flow balancing", the use of simple gravity drains and impoundments can (in the right circumstances and with adequate design) playa useful role in the mitigation of mine water pollution. For instance, the strategy to divert surface waters away from known zones of infiltration to the mine voids may be possible in some cases. This approach was used in the Muncie collapse, St John Collapse and Lytle Creek diversions by the OWRB and NRCS in 1985-86.

Where gravity drainage is not feasible because of topographic relief or pressure constraints, pumping can always be used to control and distribute mine waters where needed. Use of pumps can also be used to convey waters to active or passive treatment facilities.

Water Control Options

To control seepage flow at the Tar-Lytle confluence a combination of approaches is needed. Control of surface infiltration or recharge from surface or subsurface formations, reducing the storage space for source water from the mine void, and reducing discharge through the use of restricting flow are all techniques that will be considered pending resolution of the disposition of the chat, tailings and mill wastes currently being considered by EPA Region 6's OU4 On-going Activity. For control of the mine seepage at the Tar-Lytle confluence, an estimated flow rate of approximately 1 cfs was used in costing the two treatment options discussed below. As indicated from the 1980's flow rate for OWRB Site 4S, mean flow for the three periods slowly increased and ranged from 0.77 cfs to 1.09 cfs. The mean discharge is 0.93 cfs for the 8-year period. For preliminary design purposes, a discharge value of 1.0 cfs was used.

Flow control options would attempt to reduce, eliminate or transfer the location of this seepage in order to reduce or eliminate the need for treatment or to reduce the size (and hence cost) of the treatment needed.

Treatment Options

Two general types of treatment methods are available:

1) Active treatment methods, and

2) Passive treatment methods.

Active treatment is conventional wastewater engineering applied to mine waters. The following formal definition of active treatment emphasizes the ways in which it differs from passive treatment (see below). Active treatment on mine waters is defined as (Younger *et al*., 2002, p.271):

“Active treatment is the improvement of water quality by methods which require ongoing inputs of artificial energy and/or (bio)chemical reagents"

From the above definition, it is clear that "active treatment" embraces virtually all conventional mine water treatment technologies save those using wetlands and other "passive" unit processes (as described briefly below and by Hoskings, 2004).

The "artificial energy" referred to in the definition given above can be provided in various forms, such as electrical power for pumping, mixing, aerating etc, heat to change reaction rates or pressure to control gas-liquid exchange rates. The "reagents" used in active treatment are usually alkaline liquids or solids (e.g. calcium hydroxide, sodium hydroxide), organic polymers (for coagulation, flocculation, etc) or, less commonly, pressurized gases. One of the key advantages which active treatment has over passive treatment is the scope for precise process control: dose rates of reagents, speeds of pumps and mixers and other aspects of the treatment system can be adjusted instantaneously in response to changes in influent loadings or receiving watercourse conditions.

Furthermore, it is invariably possible to actively treat mine waters on sites much smaller than those that would be required for passive treatment, which means that active treatment will always retain an important niche for large flows where land is scarce.

Figure 15 provides a decision support scheme for deciding in a logical fashion selection of the mainstream active treatment processes.

Essentially, there is no technical limit to the quality of water that can be achieved using existing technology except for its costliness. Hence, in practice, the selection of a treatment approach to achieve the most stringent of effluent requirements comes down to an economic-environmental cost benefit analysis.

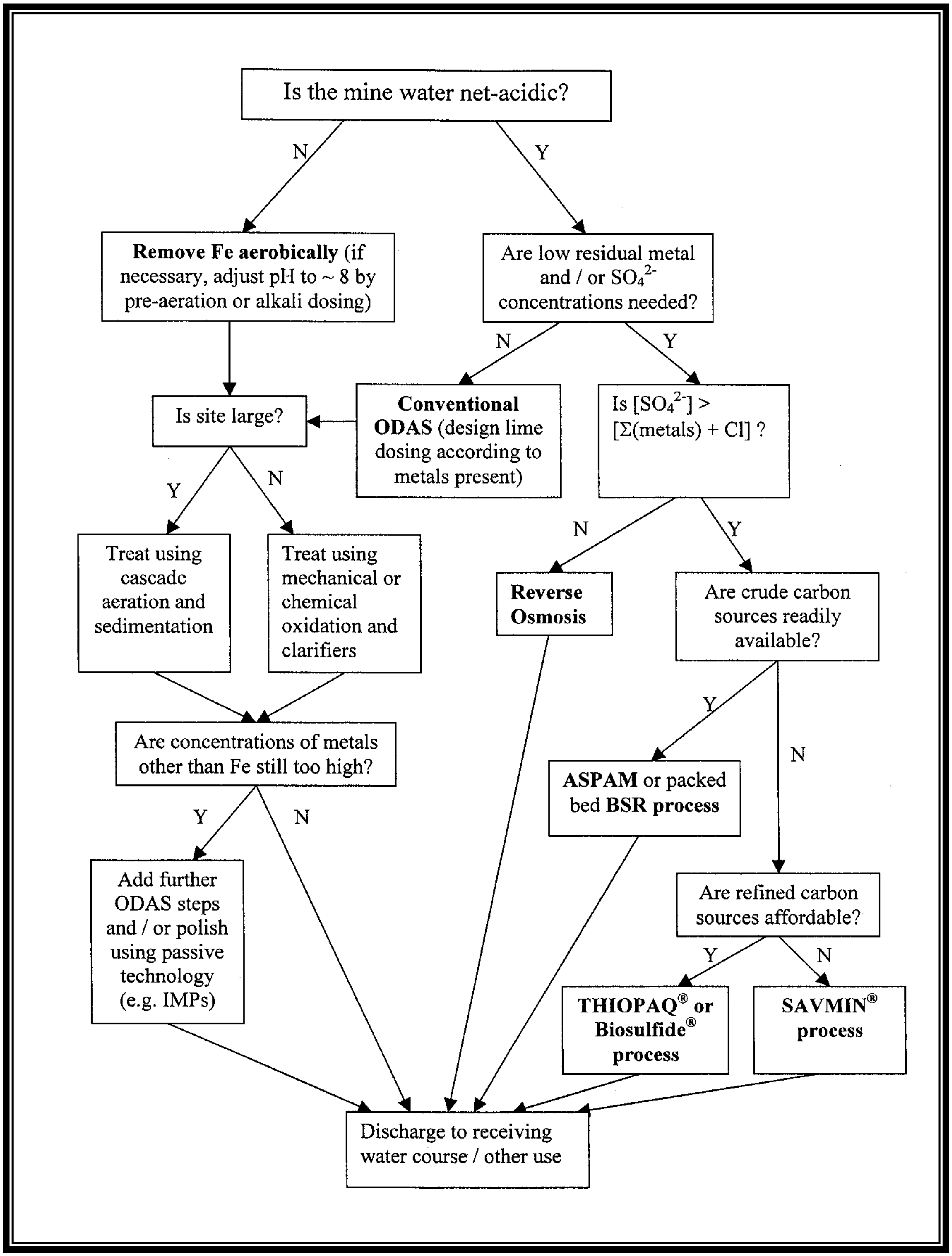

Figure 15. Decision diagram for selection of an appropriate active treatment process for a given mine water. (Adapted from Younger et al., 2002). ODAS – Oxidation, Dosing with Alkali and Sedimentation; ASPAM – Algal Sulfate Reducing Ponding process for the treatment of Acidic and Metal wastewaters; BSR process – Bacterial Sulfate Reduction; THIOPAQ® - Trademark of Paques Bio-System BV of the Netherlands: process removes both metals and sulfate from mine waters down to extremely low residuals concentrations; SAVMIN® - Trademark of MINTEK (S. Africa) Savannah Mining and the Wren Group – desalination by sulfate removal after lime dosing process; IMPs – Inorganic Media Passive systems

During the last two decades, the possibility that natural attenuating and consumptive processes can be used in mine water treatment systems has developed into a practice called *passive treatment*.

As defined by the European Union's PIRAMID R&D project (www.piramid.org):

"Passive treatment is the deliberate improvement of water quality using only naturally-available energy sources (e.g. gravity, microbial metabolic energy, photosynthesis), in systems which require only infrequent (albeit regular) maintenance in order to operate effectively over the entire system design life"

As this definition indicates, passive treatment technologies use natural materials to promote natural chemical and biological processes. Cost-effective treatment is accomplished by manipulating environmental conditions in the treatment system so particular contaminant removal processes are optimized. The principal materials utilized in passive treatment systems are locally sourced carbonate rocks and organic substrates.

Neither the materials nor the products of the vast majority of passive treatment are hazardous. It is often possible to design passive systems to operate for years (even decades) with minimal operator intervention and/or costly maintenance. Because most passive treatment systems include constructed wetlands, they may also provide wildlife habitat and can have substantial beneficial values of social and ecological natures.

Passive treatment systems are now sufficiently numerous and sufficiently mature in North America that it is possible to identify some of the key pros and cons of the technology. These are summarized in Table 11.

Table 11
Advantages and Disadvantages of Passive Treatment Systems
(Adapted from Younger *et al.*, 2002)

Advantages	Disadvantages
Low operating costs, and usually low capital costs also (at least for small- to medium-sized mine water discharges)	Passive treatment technology is still relatively new, and hence reliable expertise is still scarce
If suitably designed and well constructed, passive systems can work for long periods of time unattended	Because day-to-day intervention in treatment processes is precluded precise control of treatment effluent quality is not feasible
Passive systems can often be directly integrated with surrounding ecosystems	A large land-take is likely to be necessary for high-flow and/or highly contaminated discharges
Use non-hazardous materials	Relatively high capital (construction) costs
In many cases they will be more pleasant in appearance than active treatment systems	

The large land demand is probably the principal drawback of passive treatment. It derives from the fact that contaminant removal processes in passive systems generally occur at low or modestly high pH, under which conditions most important reaction rates (oxidation, hydrolysis etc) are considerably slower than in the artificially high-pH environments typical of conventional active treatment systems. As a result, passive systems require longer retention times and larger areas in order to achieve similar results.

The goal of passive treatment systems is to enhance the natural amelioration processes so that they occur within the treatment system, rather than in the receiving water body. As such, passive treatment can be considered an example of "enhanced natural attenuation."

Two factors that determine whether this goal can be accomplished at a given site are

1) The kinetics of the contaminant removal processes and
2) The retention time of the mine water in the treatment system.

At many sites, only a limited amount of land will actually be available, and thus an upper limit is imposed on the hydraulic retention time, which becomes a function only of flow rates. However, manipulating the environmental conditions that exist within the passive treatment system can enhance the kinetic rates of several contaminant removal processes. Efficient manipulation of contaminant removal processes requires that the nature of the rate-limiting aspects of each removal process be understood.

Passive treatment methods for mine waters fall into three main types of system:

1) Inorganic media passive systems (IMPs): This category includes a range of technologies based on the dissolution and/or precipitation of inorganic, mineral substances in surface or subsurface flow reactors. There are currently two main types of IMPs:
    a. Carbonate-dissolution based IMPs, such as anoxic limestone drains (ALDs), oxic limestone drains (OLDs), closed-system Zn removal reactors and siderite-calcite reactors for Cd and As removal , .
    b. Systems in which inorganic media provide surfaces on which mineral precipitates can accrete, such as 'SCOOFI' reactors and 'pyrolusite process' reactors

IMPs are a rapid growth area in passive treatment, and the variety of systems available is likely to expand greatly over the next decade.

2) Wetland-type passive systems. These can in turn be subdivided into three varieties, which differ radically from one another in form, function and applicability. The three types are:
    a. Aerobic wetlands (reed beds)
    b. Compost wetlands (sometimes rather inaccurately labeled 'anaerobic' wetlands)
    c. Reducing and Alkalinity-Producing Systems (RAPS). These systems were originally termed 'SAPS' (Successive Alkalinity Producing Systems) by their originators (Kepler and McCleary, 1994), but were recently re-named by Watzlaf and co-workers to better reflect their functioning.

The latter are sometimes also referred to as 'vertical flow ponds' (e.g. Demchak et al., 2001). This terminology arose from experiments with compost wetland systems in the mid-1990s, from experiments aimed at maximizing subsurface flow through compost substrates. Attempts to achieve this by horizontal flow failed because the water eventually developed flow paths on the compost surface. By contrast, systems with a vertical flow path worked. Such systems are essentially the same as 'SFBs', described below. However the term 'vertical flow pond' became established in some circles and found ready transfer to RAPS when they were invented. However, as the term 'vertical flow ponds' has very specific connotations in relation to passive treatment of sewage (e.g. Cooper, 2001), and as that connotation is synonymous with aerobic treatment, its use with another meaning in the mine water is context is probably best avoided.

3) Subsurface-flow bacterial sulfate reduction systems (SFBs). These further sub divide into two sub-categories:
   a. In-situ permeable reactive barriers (PRBs) treating contaminated ground water within an aquifer, and
   b. SFBs constructed to treat contaminated mine water discharges

Apart from their mode of construction and the locations of system influents and effluents, there is no scientific difference between these two sub-categories.

It should be stressed that the above typology is somewhat artificial, in that certain elements of IMPs technology are integral to most wetland-type systems, and there is considerable commonality in form and function between the SFBs and the subsurface portions of RAPS wetland systems. Furthermore, the treatment of any given mine water may require deployment of more than one of the above systems in series.

Combinations

Several passive treatment systems were evaluated and economics determined as options to treating the mine waters at the Tar-Lytle confluence. A constructed wetland design to treat 1 cfs is provided by Hoskings (2004) included in this Appendix.

The other system was a permeable reactive barrier (PBR) and was considered in anticipation that either less space may be required and/or be used to accommodate other spring or seep flow in the vicinity of the confluence since actually several seeps occur in the area. Further evaluation of these options will be considered when developing alternatives pending the final work plan for the On-going EPA OU4 project.

PRB System and Estimated Costs

The concept of a Permeable Reactive Barrier (PRB) is very simple (see Figure 16) a permeable medium of geochemically appropriate material is placed in the path of the polluted groundwater in the form of a 'barrier' across the flow path. As the groundwater flows through the barrier, beneficial (bio)chemical reactions take place which result in an overall improvement in water quality, so that the groundwater flowing out of the down-gradient face of the barrier is significantly less polluted than that which entered.

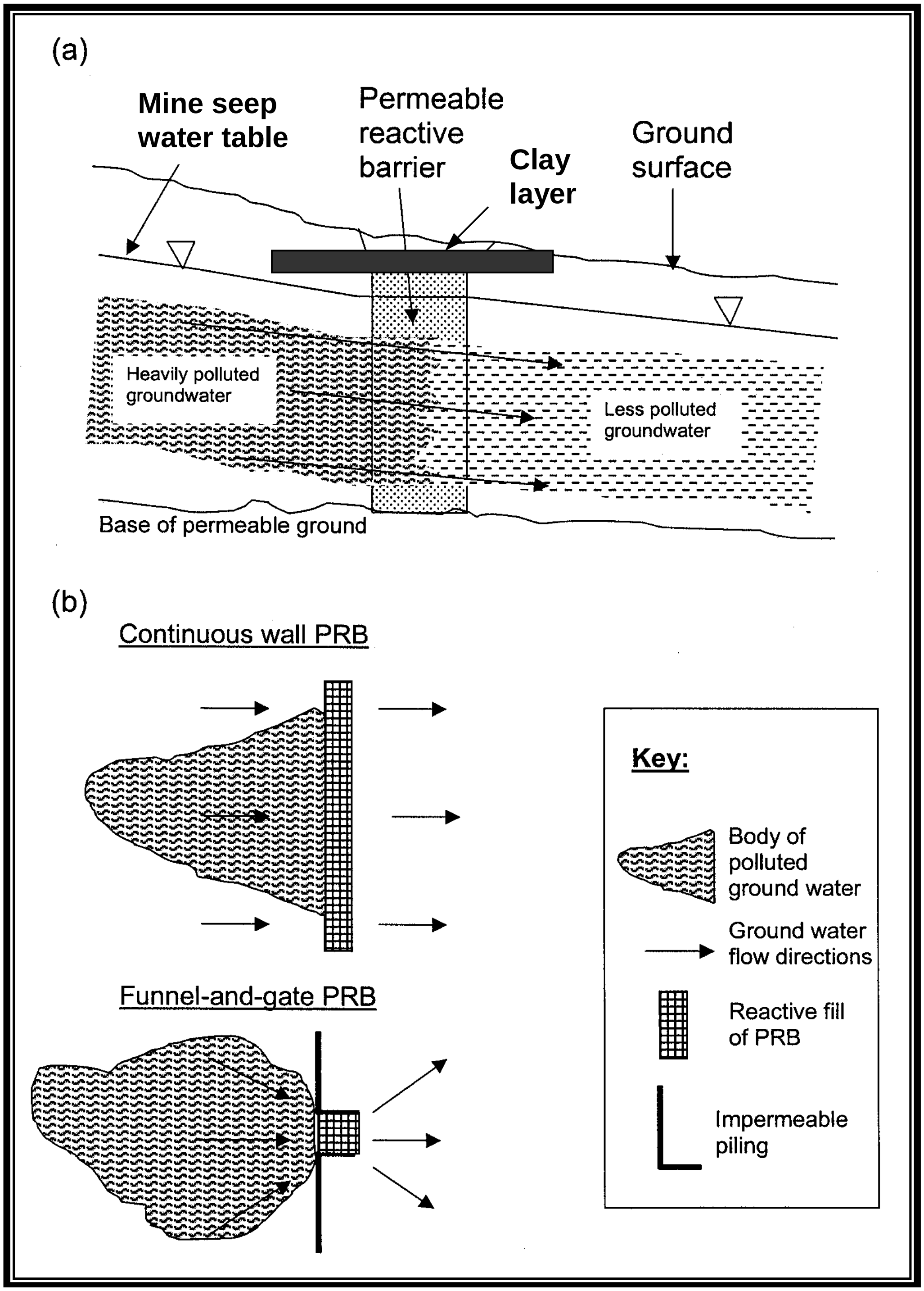

Figure 16. Permeable Reactive Barriers (PBR): (a) Generalized and simplified cross-section illustrating the basic design concept, and (b) plan view of two types of barriers, one continuous or restricted (funnel and gate). Designed based on continuous wall with multiple sand layers before and after reactive zone (see Benner et al., 1997, 1999; Waybrant et al. 1998; Benner et al., 2001)

Design Practice for PRBs: Prior investigation of the Tar-Lytle confluence to determine the mechanism and subsurface hydraulics of the seep is an essential pre-requisite for robust PRB design.

Relevant parameters required (preferably over several seasonal cycles) include:

- The hydraulic conductivity (or transmissivity) of the aquifer which are best determined by test pumping of boreholes).
- The hydraulic head distribution in the aquifer (measured using boreholes) from which groundwater flow directions can be deduced and groundwater velocities can be calculated.
- Groundwater quality as indicated by borehole and seep samples, from which the extent of the 'plume' of polluted groundwater may be mapped. Pollutant concentrations are likely to be highest in close proximity to likely 'source seep areas' (such as bodies of flooded workings, spoil heaps or tailings ponds), and to decline down the direction of groundwater flow, due to mixing and reaction with unpolluted groundwater.

Once the location and direction of flow of the body of contaminated mine water is known, and the hydraulic conductivity of the host aquifer has been determined, substrate selection will be performed.

The first step in this process is to prepare an inventory of suitable organic materials within economic haulage distance of the site. Critical issues here will be the availability of sufficient quantities to match the volume of the PRB.

In some areas, there may be so little organic material available locally that little choice may exist: all available material will be required. Usually, some element of choice will exist, and alternative media can be selected on the basis of an optimal combination of:

their capacity for promoting sulfate reduction and metal removal, and
their hydraulic conductivity.

Both of these properties can be evaluated using laboratory tests; suitable protocols for such tests are described by Waybrant et at. (1998). Laboratory microcosm tests, in which the organic material is placed in contact with a large sample of the polluted groundwater for a month or more, will allow the propensity for sulfate reduction to be evaluated. Lab permeametry can be used to evaluate the hydraulic conductivities of the various alternative barrier fill materials, as long as care is taken to impose weights on the materials in the permeameter to reproduce the effect of compaction by self-loading which will occur at depth in the field PRB (Younger *et al*., 2002). In many cases it will be necessary to create a mixture of organic material with some proportion of mineral clasts in order to obtain a suitable combination of reactivity and hydraulic conductivity (K).

Recent field and modeling studies have revealed that the target hydraulic conductivity for a PRB substrate ought to be 5 to 10 times greater than that of the enclosing aquifer (Gavaskar et at., 1998). The example mixture selected for use in the Tar –Lytle mine water PRB installation was patterned after the Nickel Rim site in Sudbury, Ontario due to similar water chemistry. The reactive barrier material composing the Nickel Rim site PBR is as follows: (Benner et al. (1997). Reactive wall material consisted of 20%

municipal compost, 20% leaf mulch, 9% wood chips, 50% pea gravel and 1% limestone (Benner *et al*., 1997; Waybrant *et al.*, 1998). Average permeability of this material was 345 m/d. (Benner *et al*., 1997).

Table 12 provides the preliminary cost estimate for this prototype design.

Table 12
Tar-Lytle Confluence PBR Preliminary Cost Estimate
based on Nickel Rim Site, Sudbury, ON, Canada

Wall length (width x depth)	Capital Costs	Annual Cost
1) 107 m (350 ft) (5m x 20m)	$ 2.7M	
2) 610 m (2000 ft) (5m x 25m)	$ 19.1M	
3) 275 m (900 ft) (5m x 25m)	$ 8.6M	
Subtotal	**$ 30.4M**	
Monitoring, Investigation & Permitting	$ 1.96M	
Flow System Inv.	$ 0.70M	
Engineering & Design Subtotal	$ 2.66M	
O&M costs		$ 0.17M
Total Costs	**$ 33.06M**	**$ 0.17M**

Gavaskar et al. (1998) describe the installation of PBRs and case examples are provided by Naftz et al. (2002).

The preliminary costs cited are tentative depending on the level of final water control implemented, length of wall needed and consideration of other treatment options pending final work plan outcome of the On-going EPA OU4 project..

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Following Figures and Table are separate sheets

Figures 5, 10a, 10b, 11a, 11b

Table 10

Table 5
Geologic units of the Boone Formation in USGS Wyandotte Quadrangle Correlated with informal letter classification of minded lithologic units in the Picher Field, OK & KS

Series	**Members of Boone Formation** **(After McKnight et al, 1970, p. 19-55)**	**Informal letter classification** **(Fowler & Lyden, 1932; Fowler, 1942)**
Overlying Chester Series (Hindsville LS, Batesville SS and Fayetteville Shale		
Meramee (Middle Mississippian)	Moccasin Bend Member	***Bed*** **B** **C** **D** **E** **F** **G** **H**
	Baxter Springs Member	**J** **K (ls w/g, ch)** **L (ch w/g)**
	Short Creek Oölite Member (karstic ls)	**M**
Osage (Lower Mississippian)	Joplin Member (karstic ls)	
	Grand Falls Chert Member (ch, 10% ls)	**N** **O** **P** **Q**
	Reeds Spring Member St. Joe Limestone Member	**R**
Underlying Chattanooga Shale **Cotter Dolomite**		

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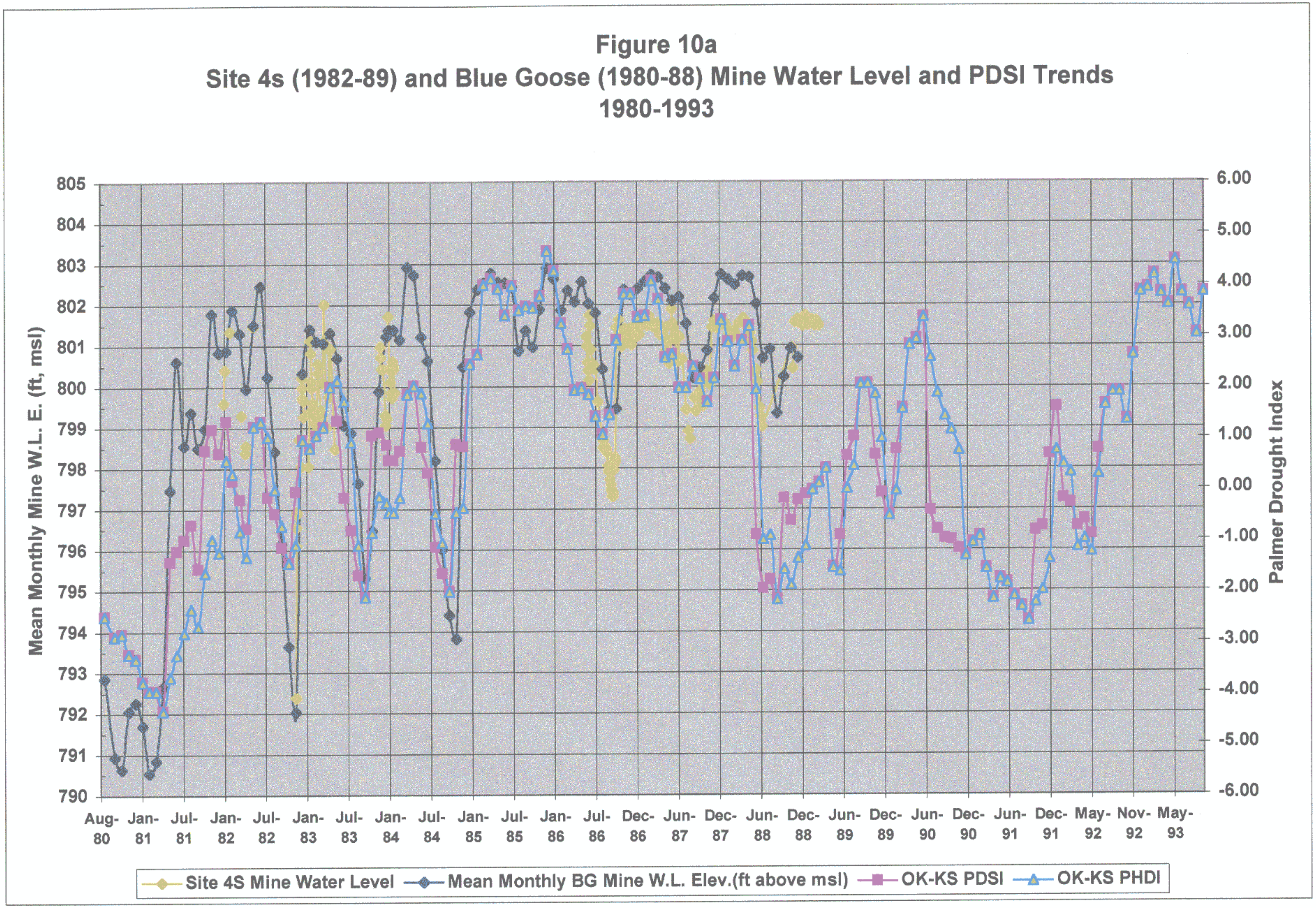

Figure 10a
Site 4s (1982-89) and Blue Goose (1980-88) Mine Water Level and PDSI Trends
1980-1993
Mean Monthly Mine W.L. E. (ft, msl)
Palmer Drought Index
805
804
803
802
801
800
799
798
797
796
795
794
793
792
791
790
6.00
5.00
4.00
3.00
2.00
1.00
0.00
-1.00
-2.00
-3.00
-4.00
-5.00
-6.00
Aug-80
Jan-81
Jul-81
Jan-82
Jul-82
Jan-83
Jul-83
Jan-84
Jul-84
Jan-85
Jul-85
Jan-86
Jul-86
Dec-86
Jun-87
Dec-87
Jun-88
Dec-88
Jun-89
Dec-89
Jun-90
Dec-90
Jun-91
Dec-91
May-92
Nov-92
May-93
Site 4S Mine Water Level
Mean Monthly BG Mine W.L. Elev.(ft above msl)
OK-KS PDSI
OK-KS PHDI

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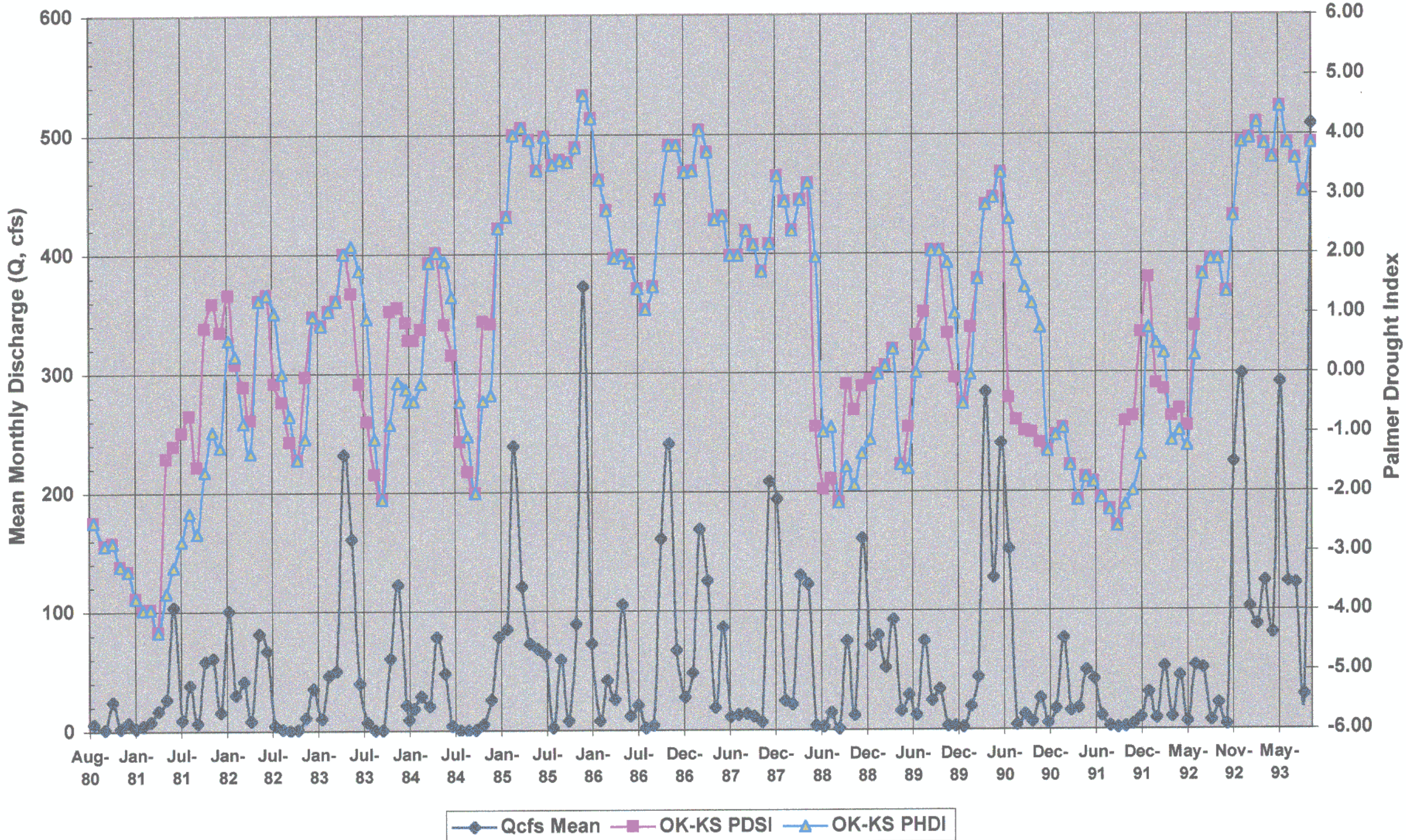
Figure 10b
Discharge and Palmer Drought Trends, USGS Tar Creek Gage, 1980 to 1993



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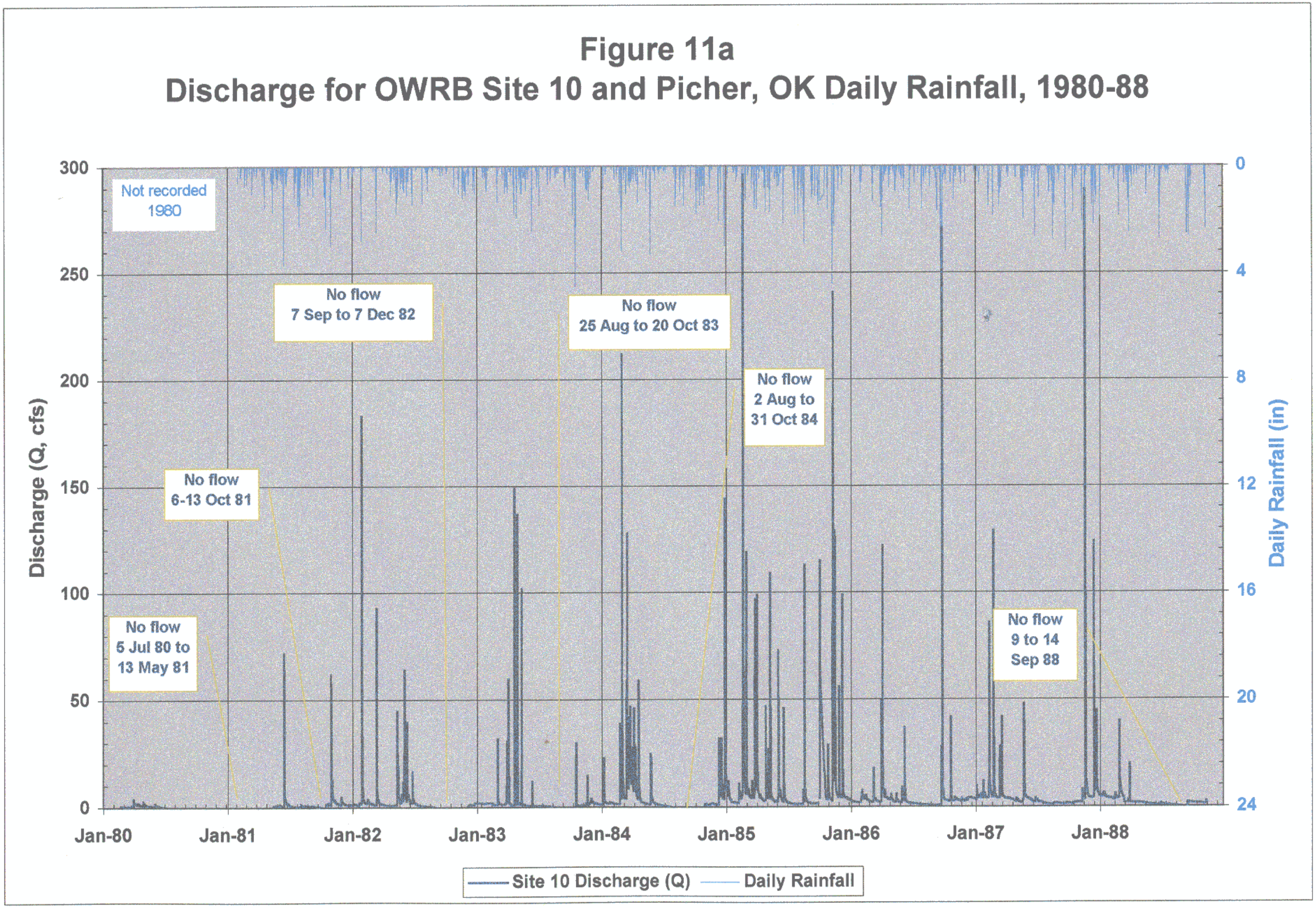

Figure 11a
Discharge for OWRB Site 10 and Picher, OK Daily Rainfall, 1980-88
Not recorded 1980
No flow 7 Sep to 7 Dec 82
No flow 25 Aug to 20 Oct 83
No flow 2 Aug to 31 Oct 84
No flow 6-13 Oct 81
No flow 5 Jul 80 to 13 May 81
No flow 9 to 14 Sep 88
Discharge (Q, cfs)
Daily Rainfall (in)
300
250
200
150
100
50
0
0
4
8
12
16
20
24
Jan-80
Jan-81
Jan-82
Jan-83
Jan-84
Jan-85
Jan-86
Jan-87
Jan-88
Site 10 Discharge (Q)
Daily Rainfall

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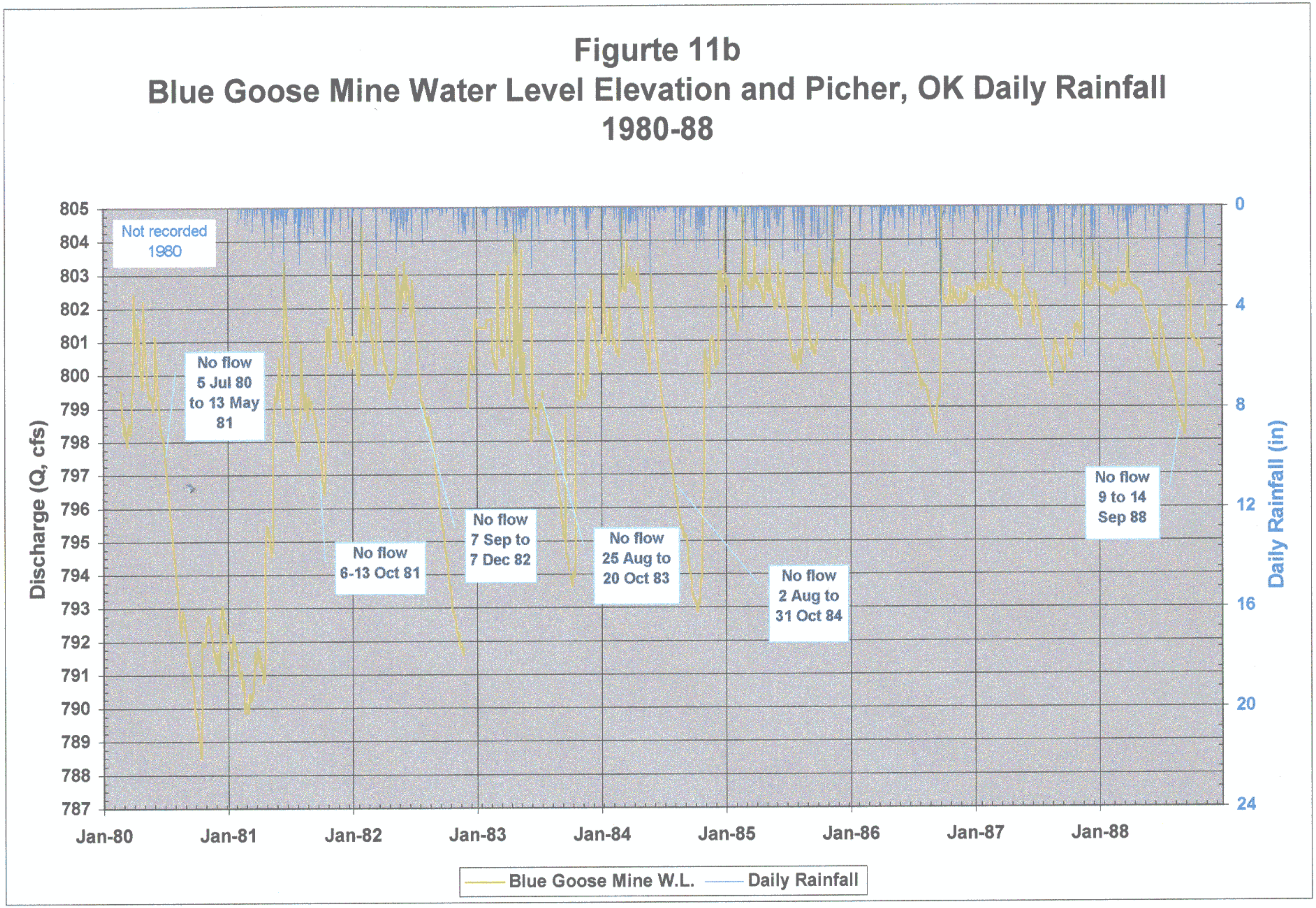

Figurte 11b
Blue Goose Mine Water Level Elevation and Picher, OK Daily Rainfall
1980-88
Discharge (Q, cfs)
Daily Rainfall (in)
805
804
803
802
801
800
799
798
797
796
795
794
793
792
791
790
789
788
787
0
4
8
12
16
20
24
Not recorded 1980
No flow 5 Jul 80 to 13 May 81
No flow 6-13 Oct 81
No flow 7 Sep to 7 Dec 82
No flow 25 Aug to 20 Oct 83
No flow 2 Aug to 31 Oct 84
No flow 9 to 14 Sep 88
Jan-80
Jan-81
Jan-82
Jan-83
Jan-84
Jan-85
Jan-86
Jan-87
Jan-88
Blue Goose Mine W.L.
Daily Rainfall

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Table 10
Water Quality Field Chemistry and Discharge Mean Values
for Three Time Periods: 1981-83, 1984-86 and 1987-89

OWRB Site	Location Name along Tar Creek	Drainage Area (sq mi)	Mean Q (cfs) per sq mi	Mean Q (cfs)			pH			Dissolved Oxygen (mg/l)			Spec. Cond. (umho/cm)		
				1981-83	1984-86	1987-89	1981-83	1984-86	1987-89	1981-83	1984-86	1987-89	1981-83	1984-86	1987-89
7	OK-KS stateline, Tar Cr headwaters	8.61	0.35	--	--	2.98	6.3	6.5	7.1	3.8	7.3	10.4	929	1118	705
4A	50 yds upstream Tar Cr - Lytle Cr Confluence (Douthat Bridge)			--	--	--	5.9	--	6.9	5.1	--	11.0	1615	--	1103
4S	Mine drillhole seep just above & east of Douthat Bridge			0.77	0.92	1.09	4.9	4.6	5.2	0.6	2.3	0.4	4326	3603	3293
4L	Lytle Cr mouth	12.06	0.05	--	--	0.57	--	--	5.6	--	--	5.1	--	--	3166
4B	100-yds below Tar Cr - Lytle Cr Confluence (Douthat Bridge)	27.50	0.65 *	--	--	--	5.6	5.5	6.1	4.1	5.8	8.1	2061	2155	1942
10	Hwy 69 Bridge	34.24	0.15	2.67	7.70	4.62	5.2	5.6	6.2	3.2	7.4	8.8	2045	2440	1994
13	Discharge from New State & King Jack collapse area			--	--	0.57	2.6		5.3	2.8		2.4	3612		2966
14 or 14S	Mayer Spring		0.37	0.36	0.40	0.43	5.4	5.1	5.5	0.6	2.2	0.4	4171	4121	4023
14B	Site 13 & 14S Confluence	1.08		--	--	--	--	5.5	5.6	--	5.0	4.7	--	3274	3220
20B	22nd St. Bridge	44.83	1.19	41.6	63.62	54.97	5.2	5.8	5.8	5.9	7.4	9.1	1605	1735	1797
20	9th St RR Bridge														
22	Hwy 10 Bridge						6.3		6.6	6.6		10.4	1120		1603
22B	Tar Cr confluence w/Neosho River	53.07					6.1		7.4	1.4		12.8	1121		528

Notes:

0.21 primarily mine seepage flow

0.05 partial Lytle Cr flow from drainage only

* Idealized estimated from regional runnoff relationships, likely high estimate since reach is expected to be losing reach

For 84-86 period, little of no field water chem data for 1985 was collected as well plugging and diversion construction activities on-going