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FORECASTING GROUNDWATER RESPONSES TO DAM REMOVAL

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FORECASTING GROUNDWATER RESPONSES TO DAM REMOVAL

By

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Masters of Science, University of Alaska Fairbanks, Fairbanks, Alaska, 2005
Bachelors of Science, University of Montana, Missoula, Montana, 2003

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PREFACE

This dissertation contains three papers describing groundwater system responses to dam removals and presents current and new methodologies that managers can use to proactively forecast and mitigate those impacts. The three papers bring together literature, data gathering, data analysis, testing, and modeling techniques that apply groundwater science to forecasting the response of groundwater systems to dam removal actions. Conceptual and numerical models developed as part of this work provide scientists, environmental consultants, regulators and managers with tools to assess the consequences of the removal of stream reservoirs on the adjacent and underlying groundwater system.

The first chapter/paper, Responses of Groundwater Systems to Dam Removal is a review paper on the connection between groundwater systems and artificial impoundments. The synthesis of materials was compiled into a general conceptual model of the effects of dam and reservoir emplacement and removal on associated groundwater systems. Additionally, a method is proposed and tested to forecast the magnitude of impacts to groundwater levels using a generalized lumped parameter approach that allows the ratio of aquifer discharges to hydraulic conductivity to vary depending on the hydrogeological setting.

The second chapter/paper, Proactive Mitigation of Domestic and Municipal Groundwater Supplies During Dam Removal Actions, Milltown Reservoir, Western Montana is an applied case study and outlines the process for mitigating water supplies from engineering actions associated with the removal of the Milltown Dam in western Montana between 2006 and 2009. This paper is a summary of four technical reports completed and submitted by these authors to the EPA (http://www.epa.gov/region8/superfund/mt/milltown/techdocs.html) outlining the groundwater mitigation processes in detail. The paper summarizes data collection and modeling approaches undertaken to provide practical forecasts of groundwater level changes. A range of forecasts are compared to completed mitigation actions and model performance is evaluated. A risk management framework is proposed and tested.

The third chapter/paper, The Role of Drawdown Data in ANN Forecasting of Water Table Responses to Dam and Reservoir Removals examines the applicability of using Artificial Neural Networks (ANNs) to forecast groundwater levels changes resulting from a dam removal. Two specific ANN models were developed and analyzed to specifically examine the need for training data inclusive of a temporary or partial drawdown. Results for the Milltown Dam removal are compared to observed water levels and results of standard numerical techniques (presented in the second paper). ANN modeling shows promise as a tool to forecast likely groundwater responses to dam removals as it requires less detailed hydrogeological data sets and is executed more efficiently than standard numerical models.


Chapter/Paper 1

Berthelote, Antony, Doctor of Philosophy, May 2013

Geosciences

Responses of Groundwater Systems to Dam Removal

Chairperson: Dr. William W. Woessner

ABSTRACT:

Dams are constructed to generate hydropower, provide water for crops and human consumption, manage floods, create navigable waterways, provide recreational opportunities, create new or enhance existing wildlife habitats, and capture contaminated sediments. Research on physical impacts of building or removing dams on local hydrology typically focuses on modifications to the river system, however, impacts to groundwater systems also occur. Dam emplacement or removal actions modify adjacent groundwater system boundary conditions and often result in a rise or fall in the underlying and adjacent water table. Unfortunately, few dam removal efforts have documented changes to associated groundwater making the formulation of impact magnitude and time forecasts challenging. This research develops descriptive and generic semi-quantitative conceptual models of the response of groundwater systems to dam and reservoir emplacement and removal. Numerical generic models are constructed using a set of dimensionless parameters and the ratio of aquifer discharges to values of hydraulic conductivity. The generic conceptual model forecasted changes in water table positions after a dam removal are then compared to observed groundwater responses during a 8.5 m high dam removal in western Montana. The simulated water table declines compared favorably with the observed declines of 0.5 to 3.0 m. Future dam emplacement and removal actions need to recognize the likely response of the local water table and, if necessary, develop pre and post dam groundwater impact mitigation measures.

KEY WORDS Milltown Reservoir; Dam Removal Mitigation And Management; Groundwater Surface Water Interactions; Groundwater Level Forecasting;

1.0 INTRODUCTION

Construction of new dams, hydropower systems, and reservoirs provide immediate benefits to many of the two billion people lacking access to electricity and seven billion expected to face water scarcity by 2050 (ICOLD, 2007; Pegg, 2004; World Commission on Dams, 2000). Pegg (2004) reported that 1500 large dams were under construction globally while, in more developed countries like the United States, dam removals were outpacing new construction. Decisions to remove dams are based on identified adverse ecological and social impacts, safety conditions associated with aging dams, and appreciation for societal values linked to healthy rivers and fisheries (American Indian Law Center, 1999; Collier et al., 1996; Collins et al., 2007; Graf, 2002, 2003, 2005; Johnson and Graber, 2002; Pejchar and Warner, 2001; Pennsylvania Organization for Watersheds and Rivers et al., 2004; Pohl, 2002; Whitelaw and MacMullan, 2002; World Commission on Dams, 2000). Recent interests in dam removals are reflected in a steady stream of dam removal articles appearing in the popular press (Babitt, 2002; Francisco, 2004; Hart and Poff, 2002; Landers, 2004; Martin, 2004; McCool, 2004; O'Connor et al., 2008; Tweit, 2006). There were over 23 news articles in the month of June 2008 alone, related to dam removals (e.g., Aun, 2008; Bouma, 2008; Caduto, 2008; Dean, 2008; Egan, 2008). Dam removal rates have been steadily increasing (Bowman et al., 2002; O'Connor et al., 2008) with 60 U.S. dams removed in 2010 alone (McClain, 2012). The last 13 years account for over 450 of the 888 large U.S. dams removed in the past century (McClain, 2012).

The impacts and changes to associated groundwater systems as a consequence of the emplacement or removal of dams and reservoirs have historically been overlooked (e.g., Doyle et al., 2003a; Doyle et al., 2003b; Evans et al., 2000a; Graf, 2003; Hart et al., 2002). In addition to the anticipated water table changes, new water table positions are also secondarily associated with extent and function of wetlands, the degree of disconnection or reconnection of groundwater with aquatic ecosystems, and impacts to surface water and groundwater quality (Aseltyne et al., 2006; Constantz, 2003; Constantz and Essaid, 2007). It is
the physical response of the adjacent alluvium dominated groundwater system to dam and reservoir construction and more specifically dam removals that is the focus of this work.

1.1 The Role Of The Dam
Starting with the first large-scale dam constructed in 5000 B.C., with the exception of the Great Wall of China, dams are the largest structures ever built (PBS, 2008). The U.S. has approximately 2.5 million small dams (less than 1.8 m high), 80,000 large dams (over 1.8 m high), and 8036 major dams (greater than 15 m high) (American Rivers et al., 1999; Bowman et al., 2002; USACE, 2008b). The U.S. accounts for nearly 20% of the 45,000+ major dams in the world today. One in four of the major dams in the U.S. are built in river locations with unconsolidated and semiconsolidated sand and gravel aquifers. Such valley settings host major aquifers (U.S. Dept. of the Interior, 2008; USACE, 2008b). Dams are operated to generate hydropower, provide water for crops and human consumption, manage floods, create navigable waterways, provide recreational opportunities, create new or enhance existing wildlife habitats, and capture contaminated sediments. Such dams also often represent an important aspect of a community’s history (American Rivers et al., 1999; Bowman et al., 2002; World Commission on Dams, 2000). Their construction commonly determined the locations of towns, industries, and trade routes. Communities benefitted from hydroelectric power, recreation opportunities, flood protection, and access to reservoir water or elevated groundwater levels for municipal and agricultural water supplies (American Rivers et al., 1999; Evans et al., 2000a; Pyle, 1995; World Commission on Dams, 2000).

1.2 Hydrologic Changes Associated With Dam Emplacement
While some studies have looked at larger-scale historical channel changes in rivers from dam emplacements (Beyer, 2005; Gregory et al., 2002; Nilsson et al., 2005; Renwick et al., 2005; Wootton et al., 1996), the impacts of dams on rivers have been most typically viewed as a surface water phenomenon involving geomorphology, surface-water hydraulics, sediment transport, fisheries, benthic and riparian ecology, as well as a plethora of aesthetic issues (e.g., Graf, 2005) (Figure 1). Changes in river conditions at dam sites include pool development, increased water depth, changes in river temperature, possible pool water density stratification, loss of light penetration due to increased water depths and turbidity, retention of nitrates and phosphates, growth of plankton and algae, and changes in aquatic ecosystems from lentic to lotic species (Baxter, 1977; Petts, 1984; Poff and Hart, 2002). In addition, it has been long documented that sediment accumulation in reservoirs will result in continual declines in water storage capacity (Dendy, 1968; Radoane and Radoane, 2005).

Downstream of a dam, a river typically reestablishes its sediment load by eroding bed and bank materials, causing incision and channel widening, and preferential transport of fine grained material (Evans et al., 2000c; Faulkner and McIntyre, 1996; Graf, 2005; Grams and Schmidt, 2005; Petts and Gurnell, 2005; Poff et al., 1997; Radoane and Radoane, 2005; Renwick et al., 2005). The inevitable result is channel embedding, which can have an adverse impact on benthic ecosystems (Petts, 1984; Petts and Gurnell, 2005). The most pervasive long-term downstream effect, however, is aggradation that results from flow regulation (Grams and Schmidt, 2005; Marston et al., 2005). The dam serves to attenuate the flood peaks that govern sediment transport in an unregulated river. The resulting reduction in sediment deposition downstream affects channel morphology, substrate, and flood stages (Chin et al., 2002; Collier et al., 1996). Downstream impacts to the river system can include alteration of the thermal structure of the river due to release of water from below the thermocline of the reservoir (Muth et al., 2000), riparian plant communities (Bayley, 1995; Doyle et al., 2005; Magilligan and Nislow, 2005; Muth et al., 2000; Petts and Gurnell, 2005; Shafrath et al., 2002), and the restriction of anadromous fish migration (Baxter, 1977; Bayley, 1995; Nislow et al., 2002; Wootton et al., 1996).
As suggested by Constantz (2003), all these previously studied vantage points possess merit, but neglect any associated physical and/or ecological responses of adjacent shallow groundwater systems. It has been long documented that increases in surface water elevation tend to increase the height of the water table in the areas immediately behind impoundments (Leopold and Maddock, 1954). A few researchers have attempted to describe and quantify likely ground water responses to reservoir pool elevation changes. Engineered pools and beaver constructed ponds are hydraulically similar to lakes and reservoirs. Thus, beaver dam studies are a natural (small to large-scale dam) analog for understanding how groundwater responds to reservoir pool manipulations. Where beaver dams span the entire valley, the main hydrologic feature will be an upstream pond that elevates groundwater levels adjacent to the pond (Butler and Malanson, 2005; Chen and Chen, 2003; Mertes, 1997; Naiman et al., 1988; Westbrook et al., 2006; Woo and Waddington, 1990). However, where valleys are unconfined, yet rivers are narrow enough to be dammed by beaver, the hydrologic effects may extend far beyond the edge of the pond (Lowry and Beschta, 1994). Guo (1997) outlined several historical and new analytical solutions for transient groundwater flow between a reservoir and a semi-infinite unconfined aquifer. He suggested that following release of water from bank storage into a reservoir, the rate of hydraulic-head change in the aquifer should decrease with distance and time. Sawyer et al. (2009) applied similar techniques to examine the effects of frequent river stage fluctuations caused by dam operation on the hyporheic zone. All of these solutions are limited to special conditions (e.g. saturated, homogeneous, and semi-infinite) seldom found in natural
settings. Modeling of groundwater where a river enters a reservoir has demonstrated connections between the river delta, floodplain terrace, reservoir stage, and groundwater levels (Rains et al., 2004). Lewandowski, et al. (2009) completed an experimental study to identify drivers of water level fluctuations and hydrological exchanges between groundwater and surface water in a saturated oxbow system. Aseltyne et al. (2006) modeling showed increases in the depth that surface-water penetrates the river bed sediments following a reservoir-stage rise as anticipated. Finally, Heilweil et al. (2005) monitored rising groundwater levels underneath and adjacent to a newly constructed reservoir atop consolidated materials.

Groundwater level changes in response to dam and reservoir construction in these settings is expected, yet, as cited above, rarely documented. Principally, the degree of groundwater level change will be a direct result of: 1) the magnitude of reservoir stage rise, 2) the rate of exchange between the reservoir, river sections, and the groundwater (losing or gaining conditions), 3) the regional groundwater conditions (constrained or unconstrained valley sediments), and 4) boundary conditions. Generally, as dams and reservoirs are constructed, the surrounding adjacent groundwater systems are likely to expand the zone of saturation raising local water tables.

1.3 **Hydrologic Changes Associated With Dam Removals**

The emerging science of observing and forecasting impacts of dam removal has been reviewed elsewhere (Doyle et al., 2003b; Evans et al., 2000b; Graf, 2003; Hart et al., 2002). Dam removals involve transient effects that introduce new concerns for watershed management and river restoration as summarized in Figure 1. (Collins et al., 2007; Hewitt et al., 2001; Pennsylvania Organization for Watersheds and Rivers et al., 2004). Major concerns typically include modifications to channel morphology (Cantelli et al., 2007; Williams and Wolman, 1984), stream and floodplain exchange processes (Graf, 2006; Kondolf, 1998), the fate of reservoir sediments (Cui et al., 2006; Doyle et al., 2002, 2003a; Evans et al., 2002; Evans et al., 2000c; Lorang and Aggett, 2005; Pizzuto, 2002; Stanley and Doyle, 2002), and the potential generation of downstream flood hazards (Roberts, 2006). Other concerns include the ecological consequence and risks to human health of released or exposed contaminated reservoir sediments (DesGranges et al., 1998; James, 2005; Shuman, 1995; World Commission on Dams, 2000). Recognized impacts to fluvial systems upstream of a reservoir include changes in the flood regime (Batalla et al., 2004; Leopold and Maddock, 1954; Poff et al., 1997; Power et al., 1996; Rowntree and Dollar, 1999), riparian ecosystems (Doyle et al., 2005; Petts, 1984; Shafroth et al., 2002), sediment budgets (e.g., Faulkner and McIntyre, 1996) or some combination of these three factors (Beyer, 2005; Bushaw-Newton et al., 2002; Graf, 2006; Lytle and Poff, 2004; Magilligan and Nislow, 2005; Poff et al., 2006). Dam removal projects also involve numerous legal issues (Bowman, 2002; Lindlof and Wildman, 2006; Nadeau and Rains, 2007).

As stated previously, dam removal science has also been historically limited to a plethora of geomorphologic studies that focus on surface processes with no connection to subsurface groundwater levels or floodplain aquifer impacts (e.g., Collier et al., 1996; Doyle et al.; Evans et al., 2007; Evans et al., 2000c; Graf, 2003; Hart et al., 2002). The same basic hydrogeological principles that pertain to increasing groundwater levels following reservoir construction (Heilweil et al., 2005) would suggest reservoir levels would decline to pre-dammed levels following a dam removal. With the exception of a few studies on beaver dams that have documented reductions in groundwater levels following removals (Butler and Malanson, 2005; Chen and Chen, 2003; Mertes, 1997; Naiman et al., 1988; Westbrook et al., 2006; Woo and Waddington, 1990), only a small number of studies have addressed how dams alter the rates and locations of near surface hyporheic exchanges, with little emphasis on any connection to the local water table position (Alexander and Caissie, 2003; Dahm et al., 1998; Hayashi and Rosenberry, 2002; Hendricks and White, 1991; Palmer, 1993; Puschkul et al., 1998; Stanford, 1998; Stanford and Ward, 1993; Valett et al., 1990). The United States Army Corps of Engineers’ Dam Removal Research Office, which is responsible for all U.S. dam removal oversight; American Rivers, a nonprofit organization dedicated to the protection and restoration of North America's rivers and possibly the leading authority on dam removals; University of California’s Clearing House for Dam Removal Information; and Oregon States’ Dam Removal Listserv all report no information covering reservoir-groundwater level linkages other than those by these authors (American Rivers, 2008; CDRI, 2012; Oregon State, 2008; USACE, 2008a).

A central Vermont village experienced a water shortage following a dam removal (Pyle, 1995) which resulted in the Federal Energy Regulatory Commission (FERC) allowing the shallow groundwater supplied
village to buy control of a second dam scheduled for decommissioning to prevent its removal (Graf, 2002). More recently, unforeseen "dry" wells occurred following the 39 m Condit Dam Removal (Oct 2011) on the White Salmon River in Washington (Learn, 2011). The homeowners are asking the PacifiCorp Power Company who owned and removed the dam to assist in mitigating the loss of local groundwater supplies. The associated Condit Dam Environmental Impact Statement (EIS) stated “Significant unavoidable adverse impacts were not identified with respect to groundwater” (Sandison, 2010). Alternatively, the EIS for the upcoming 2.75 m Finesville Dam removal on the Lower Musconetcong River in New Jersey states that a “Potential drop in water table may result in lower water levels in some wells” (USDA, 2010). The EIS for the Gold Ray Dam Removal on the Rogue River in Oregon stated “Wells upstream of the dam could be affected by lower water levels” (NMFS, 2010). During the 6 m Wadsworth and 4.4 m Sterling Lake Dam removals in the Mantua Creek Watershed in New Jersey, several residents noted that the water table decreased following declines in reservoir levels, and they expressed concern about the need for well mitigation following the complete dam removal (Wyrick et al., 2009).

The proposed removal of the Rodman Dam on the Ocklawaha River in Florida was accompanied by a suggested research plan (the first mention of this type of research) to determine the impacts of complete drawdown from dam removal on groundwater levels, including effects on water levels in residential wells, discharge from springs and water levels in nearby lakes and wetlands (Shuman, 1995). When the initial water study was completed, it incorporated a numerical groundwater model that was used to determine sustainable groundwater pumping yields following the dam removal, but did not specifically address drawdown induced groundwater level impacts and/or impacts to hyporheic exchange and their effects on ecological system responses (Hall, 2005).

Current EIS statements and public reactions are highlighting potential negative impacts to groundwater systems from dam removal actions; however, rigorous proactive research and planning for groundwater impacts are conspicuously absent. This may necessitate costly mitigation and litigation (Bowman et al., 2002; Bowman, 2002). Legal issues of liability over well production losses are complex and often involve conflicting regulations from multiple agencies (local, tribal, state, and Federal). Attempts to plan for and mitigate groundwater, channel and riparian system impacts from dam removal actions can be accomplished if anticipated (Hart et al., 2002; Hart and Poff, 2002).

1.4 Challenges Of Forecasting Groundwater Impacts From Dam Removal
Hydrogeological science provides a clear framework within which to develop cause and effect models of groundwater level response to dam and reservoir construction and removal actions. However, impacts at both the local hyporheic (e.g., Alexander and Caisse, 2003; Dahm et al., 1998; Hayashi and Rosenberry, 2002; Hendricks and White, 1991; Palmer, 1993; Pusch et al., 1998; Stanford, 1998; Stanford and Ward, 1993; Valett et al., 1990), and valley wide (regional groundwater) scale have rarely been measured (e.g., Heilweil et al., 2005) or modeled. Constantz and Essaid (2007) attempted to highlight how changing reservoir and river management could impact downstream water supplies in California. They used a MODFLOW model to predict generic responses of downstream pumping water levels following a dam removal in a groundwater basin with altered groundwater recharge resulting from the conversion from perennial stream leakage to ephemeral stream leakage.

2.0 RESEARCH PURPOSE
This research was designed to develop and test a conceptual model that will provide a framework within which changes to groundwater levels in response to dam removals can be formulated and evaluated. It is tested using generic settings and develops a relationship that can be used to develop initial assessments of water level impacts. A case study is also presented where observed groundwater level changes are compared to the proposed conceptual model groundwater responses.

3.0 CONCEPTUAL GROUNDWATER RESPONSE MODEL
Developing a conceptual model for a wide range of stream and hydrogeological settings is desirable; however, such an approach involves a large number of variables. We developed a conceptual model that was constrained to represent natural, dammed and restored river reaches located in upland confined, semi-confined and un-confined or broad valley settings for both high and low water table systems (Figure 2). The high water table scenario includes a floodplain/valley water table that is generally higher than the river...
channel stage (overall gaining stream), and the low water table scenario features a water table that is generally lower than the river stage (losing stream). In general, a confined valley system will have a narrow floodplain that contains sediments that are coarse grained, with a steep upland topography, and a high longitudinal riverbed gradient. Dams placed in such settings often fill the river valley. Whereas, the wider un-confined floodplain setting generally has a much lower relief, floodplain sediments include larger quantities of finer material, and the longitudinal riverbed gradient is more gradual. Dams constructed in these settings are often wider than higher and partially fill the floodplain. The relationships between riverbed/reservoir sediment budgets, surface water distributions, groundwater levels, and surface water exchanges are illustrated for each stage of the river evolution.

Dams impound rivers. The reduction in longitudinal riverbed gradients and increase in cross sectional wetted area at the reservoir-river transition slows water velocities and begins aggrading sediments (if they are present in the system). Though the aggradation of these new transition zones with fine sediments may potentially reduce riverbed leakage to the adjacent groundwater, an increase in wetted surface area, and rise in river/reservoir stage will generally act to locally raise the associated groundwater levels. The reservoir head causes seepage from the reservoir and beneath the dam that discharges to the river reach downstream of the dam. The potential below dam reduction in the channel sediment budget degrades and coarsen the river channel, in some settings, and may lead to an increase in channel bed leakage extending far downstream (a factor that may also cause some increase in below dam groundwater levels). Channel incising may also induce additional groundwater discharge and effectively lower associated groundwater levels in some settings.

Immediately following a dam removal, a portion of the accumulated reservoir sediment will migrate downstream and aggrade or embed the channel causing a decrease in groundwater surface water exchange and a change in the channel conditions that can increase the potential for downstream flooding. This may also temporarily reduce the downstream groundwater levels (or increase them if groundwater is discharging to the channel). Changes in the local hyporheic exchange process, sites, timing and magnitudes is also likely to occur.
4.0 A GENERIC BOX MODEL TO ASSESS GROUNDWATER LEVEL RESPONSES

It is apparent from the conceptual model presented above that a number of groundwater responses can occur from dam and reservoir construction and removal. In an attempt to simplify the wide range of factors influencing groundwater level responses a generic box model was developed. This model assumes steady state, isotropic and homogeneous and rectangular boundary conditions. Using the relationships expressed in the groundwater flow governing equations, a number of hydrogeologic scenarios can be assessed. Proceeding with the understanding that for any given conditions, the primary factors driving the relative position of groundwater levels beneath a river channel atop unconsolidated materials are found in the Darcy flow equation:

\[ Q = K i A \]

where

- \( Q \) = Volumetric Flow Rate (\( L^3/t \))
- \( K \) = Hydraulic Conductivity (\( L/t \))
- \( i \) = Hydraulic Gradient (\( L/L \))
Under steady state conditions, for the same values of \( i \) and \( A \) variations in the value of \( K \) will yield specific groundwater flows, \( Q \). It is also evident that corresponding groundwater head values, \( h \) (where \( i = \Delta h/\Delta l \) and \( \Delta l \) is constant) will also vary under different ratios of \( Q/K \). Thus a set of generic models can be developed to evaluate how the groundwater head distribution will vary within a number of hydrogeologic settings expressed solely by ratios of \( Q/K \) (Figure 3). A steady state three layer homogeneous unitless groundwater model was designed using the MODFLOW graphical user interface program available from Environmental Simulations Group (Groundwater Vistas 5).

The dimensionless hydrogeologic setting model included three vertical layers consisting of two base layers each 50 units thick with a 250 unit thick surface layer. Model dimensions were 5000 units wide by 25000 units long with an overall depth of 350 units. The model consisted of 187,500 individual 100 unit\(^2\) cells. The MODFLOW river package was used to place a river down the central transect at a slope of 0.004, A 5 unit river stage, 2 unit riverbed thickness, and 10 unit riverbed hydraulic conductivity were designated for each river cell. In the center of the model at cell location 100 (upstream) from the left boundary, a dam was represented with a 1 cell thick wall \((K = 10 \text{ units})\) in layer 1 that extended 10 cells to either side of the river \((2000 \text{ units total width})\). The reservoir was represented by river cells behind the dam extending upstream 50 cells \((5000 \text{ units})\). These reservoir river cells maintained the same architecture as the central river cells with the reservoir stage \((\text{depth of reservoir pool})\) varying from 8 units at the upstream end to 28 units immediately behind the dam. The horizontal to vertical hydraulic conductivity ratios were set to 10:1 for all model cells. The downstream face of the model was represented with a general head boundary \((\text{where the water leaves the model})\) and the upstream face was set with a constant flux boundary by using a series of wells producing a cumulative volumetric flow rate of 11,920,000 \((\text{length}^3/\text{time})\) \((\text{groundwater entering the model})\). Remaining boundaries were no flow. Proportional hydraulic conductivities values were set to represent \( Q/K \) ratios of 100, 1,000, 3,000, and 10,000. A compilation of model input files is located in Appendix 1A.

Using this framework if a meter scale was applied, it would represent a set of conditions where the total aquifer thickness was 350 m, the reservoir has a head of 28 meters at the dam and covers an area of 10 km\(^2\). The rectangular block of sediments (aquifer) cover a valley that is 5 km wide and 25 km long. The inflow to the model \((Q)\) is in m\(^3/d\) and \(K\) is in m/d. \(Q/K\) ratio would be in units of square meters. Alternatively, if a decimeter scale were applied the total aquifer thickness would be reduced to 35 m, the reservoir head would be 2.8 m covering 0.1 km\(^2\), and the block of sediments would represent an area that is 0.5 km wide by 2.5 km long. \(Q/K\) ratios would be square decimeters.

**Figure 3** Illustration of a simplified homogeneous unitless (any consistent length and time unit could be used) three layer box model of a valley river system containing a dam and associated reservoir. This model was constructed to generally approximate the Milltown Reservoir valley in Western Montana (Berthelote et al., 2007). It was built in Groundwater Vistas using MODFLOW with both the reservoir in place and removed in order to conceptually examine groundwater impacts associated with various aquifer conditions \((Q/K\) ratios).
The relative difference in head between the river stage/reservoir stage and the groundwater for hydrogeologic settings where the flow to hydraulic conductivity ratios (Q/K) are 1,000, 3,000, and 10,000 are presented in Figure 4. Under undammed conditions or post dam removal settings (Figure 4A), modeling suggests that when the Q/K ratio is large the water table is at or near the stream stage creating a gaining stream reach in which groundwater discharges to the stream (Figure 2A conditions). In contrast, when Q/K ratios are small the water table is lower than the stream channel stage and the river channel leaks water into the groundwater system (losing channel Figure 2C). When a dammed setting is represented by the model (Figure 4B), larger ratio values appear to extend gaining river reaches farther upstream with overall higher water table positions throughout the system (leaking of reservoir water into the groundwater). Only the lowest ratio tested suggests the water table elevation proximal to the reservoir would be lower than the river channel elevation (losing stream) below the dam.

Figure 4 Illustration of the modeled groundwater head profile coincident with the river channel (Figure 3). Groundwater positions relative to the river channel elevation in each cell are presented for Q/K ratios of 1,000, 3,000 and 10,000 In A, the undammed scenario, as you increase the Q/K ratio by either increasing the flow or decreasing hydraulic conductivity the groundwater level increases. Conversely it decreases with a reduction in the ratio. In B, the dammed scenario, the area proximal to and downstream of the reservoir is highly influenced by the recharge from the reservoir.

The box model results (Figure 4) illustrate steady state conditions with and without a dam and reservoir in place. The figure can be used to illustrate the groundwater transition from free flowing stream conditions (A) to a dammed scenario (B), or the response of a dammed system (B) to a dam removal (A). Though these modeling results represent stabilized steady state settings it is realized that in some cases the dam removal process is not instantaneous. Large scale dam removals are often done in stages. As a result, in some settings, the river and groundwater systems may take months to years to fully revert to steady state conditions following a dam removal. The results of the generic modeling will be evaluated by comparing them to the observed groundwater response to the 2006 to 2009 removal of the 8.5 m high Milltown dam and reservoir, western Montana (Berthelote 2013).
5.0 case study - milltown reservoir

The 8.5 m high Milltown Dam construction was completed in 1907 at the confluence of the Clark Fork and Blackfoot rivers. Over the next 100 years, the Milltown reservoir filled with mining and smelter wastes from the Butte and Anaconda area located 140 km upstream. The Milltown Reservoir was designated a CERCLA (EPA Superfund) site in 1983 as water seeping from the reservoir sediments recharged the adjacent coarse-grained aquifer and contaminated local wells with dissolved arsenic, iron and manganese (ARCO 1992; Harding Lawson Associates (HLA) 1987; Moore and Woessner 2002; Udaloy 1988; Woessner et al. 1984).

The river systems and reservoir are located in a semi-confined mountain valley setting in which the valley floor sediments (6 to 60 m thick) are dominated by fluviatil sand, gravel, cobbles and boulders deposited by the ancestral Clark Fork and Blackfoot Rivers. Additional coarse grained sediments were deposited from receding Glacial Lake Missoula floods. These sediments are bounded by steep mountain boundaries composed of argillite, quartzite and limestone metasediments of the Precambrian Belt Series (Gestring, 1994). Residents located adjacent to and proximal to the reservoir utilize the prolific unconfined valley aquifer (with water table depths of 2 to 35 m below land surface) for all domestic and municipal water supplies (Berthelote and Woessner, 2009).

Initial pre-dam removal field data revealed the rivers and groundwater systems formed a complex hydrogeological system. The highly conductive aquifer (range of hydraulic conductivity from 90 to >27,000 m/d) was recharged from four sources: 1) a perched and leaking (losing) Blackfoot River arm of the reservoir; 2) most all of the losing Clark Fork River channel from just below the dam to Hellgate Canyon (vertical riverbed hydraulic conductivities range from 0.4 to 12.8 m/d and leakage rates from 68,000 to 284,000 m$^3$/d); 3) lateral underflow in the Clark Fork and Blackfoot river valley and; 4) limited recharge from the mountain boundaries. Aquifer discharge occurred in a short gaining reach directly below the dam (300m), as underflow through the valley aquifer located in Hellgate Canyon to the west (~283000 m$^3$/d ), and locally in gaining stream sections of the Clark Fork River above the reservoir. Generally, groundwater flowed towards the reservoir area from the upper Clark Fork River valley and converges just above the reservoir with the groundwater entering at the mouth of the Blackfoot River canyon, then flows northwest down valley (Figure 5). Valley widths are approximately 1 km +/- 0.5 km.

Mean annual flows for the Clark Fork River below the dam (USGS station # 12340500) ranged from 70 m$^3$/d to 86 m$^3$/d during the study period. The aquifer flow rates are measured in tens of meters per day due to the coarse grained nature of the sediments, porosities of ~20% and prevailing groundwater gradients.
In 2004 the U. S. EPA, the State of Montana and other stakeholders decided to remove the 8.5 m high Milltown Dam and 1.9 of the 5.0 mcm (million cubic meters) of contaminated reservoir sediments. The goals of the removal efforts were to restore groundwater quality, provide fish passage, and return the two rivers to a natural, free-flowing state (River Design Group et al., 2008; Westwater Consultants et al., 2005). The Milltown Dam and the associated reservoir were removed during the period of 2006 to 2009 (Envirocon 2006; Westwater Consultants, River Design Group, and Geum Environmental Consulting 2005; Environcon 2006). This estimated $100+ million remediation/restoration project required three drawdowns starting in 2006 (3.5 m in March 2007, 3.5 m in March 2008, and 1.2 m in April 2009) that correlated with engineering tasks prior to reaching the final free flowing state in 2009. Before the dam removal process formally began, project engineers initiated a 3.5 m temporary drawdown in November of 2005 to examine the submerged portion of the dam. It was immediately observed that groundwater levels in some wells adjacent to the reservoir declined and a few shallow domestic wells became inoperable. These conditions resulted in the initiation of an expanded water level monitoring network and the construction and calibration of an industry standard three dimensional numerical groundwater model (Berthelote et al., 2007; Berthelote et al., 2010). The monitoring network consisted of 78 wells located at 56 locations. Groundwater levels at 22 wells were recorded at intervals no greater than 60 minutes using Solinst® continuous water level recorders (recording pressure transducers corrected with readings from a separate Solinst® barlogger). The remaining wells were measured monthly using an electric water level tape.

The State of Montana and U.S. EPA implemented a well replacement and mitigation program that attempted to proactively mitigate water supply issues prior to likely well failures. A comparison of the observational field data and the previously described conceptual and box models follows.

6.0 RESULTS

The Milltown Reservoir site would be classified as a semi-confined valley high water table system (Figure 2 as a B2 type scenario) from the middle of the reservoir upstream. The remaining valley below the Blackfoot river arm is represented by a lower water table E2 type scenario. This difference is caused by the widening and deepening of the valley sediments at the river confluence and is concurrently dependent on the presence of high aquifer hydraulic conductivities.

The observed groundwater level data presented in Figure 6 suggests portions of the groundwater system declined 2 to 3 meters as a consequence of the reservoir and dam remediation activities. This conclusion is based on comparing the observed March 31, 2006 water table levels with the observed post dam out March water levels. March water levels are consistently the lowest annual groundwater levels in this groundwater system. The computed spatial distribution of the observed changes during the low water table period was considered the best representation of the total impact to the groundwater levels from the removal activities as observed in 2010. The water level declines were not limited to the reservoir area but extended at least 6 km downstream and 2 to 3 km upstream. The magnitude of change was dependent on the proximity to the reservoir and groundwater flow directions.
Figure 6  Map showing the decline in meters of the March low water table position from March 2006 to March 2010. Wells A, B, and C illustrate transient observation data at locations downstream at, and upstream of the reservoir, respectively. The actual full pool wetted reservoir area was interpolated as no well data were available in this area (pre-dam removal wells were lost to construction activities and no 2010 head differences were available). However, the magnitude of this interpolated data (directly under the reservoir bed) is consistent with anticipated declines underneath the removed 8.5 m deep reservoir.

Milltown Site was box modeled in units of feet with the dimensions, boundaries and gradients shown in Figure 3. The dam was 28 ft high, the reservoir about 2000 ft by 5000 ft, the aquifer thickness 350 ft and Q/K ratios ranged from 500 ft² to 2000 ft². Generally, Q/K ratios average approximately 900 ft² below the dam (wells A and B) and 1300 ft² above the dam (well C). Hydraulic conductivity values in the Milltown Aquifer site cover four orders of magnitude. A model was constructed using “average” site conditions with a Q/K ratio of 1,100 ft². Then, in an attempt to better represent the variation in Q/A ratios known to occur, a hybrid model was created by combining the results of two additional constant Q/K value models (Figure 7). For the hybrid model, groundwater level changes above the dam area were derived from completing the modeling with a constant Q/K ratio of 1,300 ft². Results from a second set of model runs using a constant Q/K ratio of 900 ft² was used to represent conditions below the dam location. Both the use of average Q/K ratios and the combined hybrid model yield similar results and are inline with observations (Figure 7B).
Study site heterogeneities often make it difficult to develop generic approaches for assessing responses of groundwater systems to changes in stream or reservoir water levels. Hydrogeological conditions at the Milltown Dam site were generally represented by the generic model developed for this research. The simplified box model (Figure 4) generally produced the expected outcomes when a site-based average Q/K ratio was uniformly applied. The simplified Q/K block model forecasted groundwater level declines that compared favorably with observed changes, within about 1 m.

The hybrid approach presented here simply combined results from two models that used different constant Q/K ratios. These modeling results also closely approximated observations. Where conditions at Milltown may be somewhat unique as both flows and hydraulic conductivities are high, modeling suggests that in aquifers impacted by a dam and reservoir where the area is dominated with gaining stream reaches, impacts may not be as large as observed at Milltown. In contrast, at sites where dams are removed, Q/K ratios are small, and streams are losing groundwater changes may be larger than observed. In addition to the uncertainties related to site conditions represented in any model used to forecast possible groundwater level responses from a dam removal, isolating the response of the groundwater system that is solely attributed to the dam removal action is not as straightforward as might be thought. As the model forecast produces predicted changes under steady state conditions, in reality during the period of dam removal variations in river flows and natural or induced groundwater recharge and discharge may produce additional groundwater changes. These conditions may transiently reduce or enhance the groundwater response to a dam removal operation. For example, at the Milltown site, a series of drought years prior to removal and both normal water budget and drought conditions during the three year period of dam removal influenced final observed water levels. Groundwater systems associated with reservoirs that are principally recharged by the reservoir and river systems are most likely to have the largest changes in groundwater levels during dam removals. It is recommended that environmental and hydrogeologic monitoring prior to, throughout, and following future dam removals will provide needed data sets that can be used to develop predictive groundwater level change models for future dam removals. When extensive pre-removal hydrogeologic data sets are not available, the development of general box models such as the ones developed here can provide managers with the general magnitudes and distributions of likely groundwater responses. Future groundwater level mitigation activities associated with dam removals will undoubtedly benefit from effective forecasts of groundwater level changes.

7.0 DISCUSSION

8.0 CONCLUSION

Project managers need to be keenly aware of the connection between dam removal activities and the likely magnitude and distribution of groundwater level changes. This research derived a general conceptual
model of the effects of dam and reservoir emplacement and removal on associated groundwater systems. Relationships between valley width, sediment scour, aggradation, and surface water groundwater interactions are highlighted for each physical domain in a high and low water table setting (Figure 2). The conceptual model tested used generic settings that showed that groundwater Q/K relationships are inversely proportional to the likely magnitude of groundwater level impacts from dam removals. A case study is presented and compared to the proposed conceptual model of groundwater responses. For the Milltown Dam removal in western Montana, hydrographs and spatial extrapolation of observed groundwater level declines indicate significant overall changes in the water table position since reservoir remediation. Observed declines in groundwater levels following the dam removal were 2 m, 3 m, and 0.5 m at wells located below the reservoir, at the dam site, and above the reservoir, respectively. The observational data indicated that the magnitude in change was dependent on the proximity to the dam, and in this case, extended over 6 km downstream of the dam location and 2 km upstream. Application of generic box modeling that used both average Q/K site conditions and a hybrid approach produced similar magnitudes and patterns of groundwater level declines. Forecasting likely groundwater changes prior to dam removal activities will provide managers with information needed to initiate groundwater level mitigation planning.
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Appendix 1A  Box model Input Files for different Q/K ratios (100, 1000, 3000, 10000) for dammed and natural systems (Files available in digital format only)
Chapter/Paper 2

Proactive Mitigation of Domestic and Municipal Groundwater Supplies During Dam Removal Actions, Milltown Reservoir, Western Montana

ABSTRACT:

It has recently been recognized that reservoir recharged groundwater supplies are increasingly being threatened by dam removals. Managing groundwater system responses to dam and reservoir removals and the resulting economic consequences require time effective mitigation strategies informed by groundwater level forecasting. In western Montana project managers of the recent 8.5 m high Milltown Dam and the contaminated reservoir sediment removal implemented a well replacement mitigation strategy that attempted to proactively mitigate groundwater supply impacts. At the initiation of dam removal activities, an extensive groundwater and river system monitoring network was established to observe water level changes. A suite of multi-layer three dimensional finite difference groundwater models calibrated to both historical data and observed groundwater responses to staged reservoir drawdowns provided groundwater level forecasts of post-dam out conditions. These data were inputs to a decision tree that identified wells needing replacement, lowering of pumps, further data collection, or were considered not at risk. Uncertainty was evaluated using sensitivity analyses and resulting alternative conceptual models. Observed results showed water table changes were up to 3 m near the reservoir and impacts extended 6 km downstream and about 2 km upstream of the reservoir. Model forecasts of groundwater level changes were greater than post dam removal observed levels, as expected, because it is likely that natural variations in annual recharge and river leakage that occurred during the dam and sediment removal period did not match low water table position forecasts. Risk tree analyses showed up to 115 wells were at risk. Proactive mitigation included the construction of 80 new wells and lowering of 20 pumps. Most future dam removal projects would benefit from groundwater impact analyses and proactive groundwater supply mitigation as needed. This process should include both observations of changes and pre-dam removal forecasting of groundwater responses. Forecasts should then be assessed and actions taken based on a logical project based decision tree such as the one developed here.

KEY WORDS Groundwater Level Forecasting; 3D Numerical MODFLOW Modeling; Milltown Reservoir; Dam Removal Mitigation And Management; Groundwater Surface Water Interactions

1.0 INTRODUCTION

There is increasing interest in removing dams in the United States to remedy adverse ecological impacts, eliminate risks associated with the deteriorating conditions of aging dams, and address societal pressures to restore rivers to more natural settings (Babbitt, 2002; Farinacci, 2009; Hart et al., 2002; Landers, 2004; O’Connor et al., 2008). Concerns have been raised that emptying reservoirs as dams are removed will have local to regional impacts on domestic, municipal, and agricultural groundwater availability and use. Until recently, dam removal projects were commonly planned without consideration of the likely corresponding changes in groundwater conditions (Berthelote, 2013Chapter 1). Previous evaluations of water table impacts resulting from dam removals have been limited to comparative studies of small scale beaver dam removals, a model that may not adequately represent groundwater responses for large scale dam removal actions (Butler and Malanson, 2005; Chen and Chen, 2003; Mertes, 1997; Naiman et al., 1988; Westbrook et al., 2006; Woo and Waddington, 1990). It is encouraging that recent pre-dam removal environmental assessments for the 2.75 m high Finesville Dam (Musconetcong River in NJ) (USDA, 2010) and 11.2 m Gold Ray Dam (Rogue River in OR) (NMFS, 2010) report the possibility of well failures. However, “after the fact” well mitigation is more the norm. As an example, the 39 m high Condit Dam (White Salmon
River in WA) Environmental Impact Statement (EIS) predicted “... no significant unavoidable adverse impacts...” from groundwater level changes (Sandison, 2010). However, well mitigation was being considered to replace multiple failed wells following the dam removal (Sandison, 2010). Though current EIS statements and public reactions are highlighting potential negative impacts to groundwater systems from dam removal actions, rigorous proactive research and planning are conspicuously absent, a condition that may result in costly mitigation and litigation in some settings (Bowman et al., 2002; Bowman, 2002).

This research was designed to assess a process used to forecast groundwater level changes prior to and during a dam removal, and evaluate if mitigation strategies executed prior to the final dam out scenario were effective in reducing actual and perceived impacts to associated groundwater supplies. The following hypothesis was tested:

**Hypothesis:** Model forecasts of the water table position resulting from planned sequential drawdowns and final dam removal will identify at least 70% of the water supply wells requiring groundwater level response mitigation.

We evaluated this hypothesis during the staged 8.5 m high Milltown Dam removal project that occurred between 2006 and 2009. The dam created Milltown Reservoir located in the semi-confined alluvial valley of the Clark Fork River in western Montana. Just prior to the initiation of the first phase of planned reservoir drawdowns, a groundwater monitoring network was initiated. Numerical groundwater modeling was developed to forecast the response of the water table to reservoir drawdowns and final dam out conditions. As staged removal plans were executed, a decision tree populated by modeling forecasts was created that assessed the risk of impairment of valley domestic and municipal water supplies. The processes used to derive and assess observations and model results within a risked based decision tree are the subject of this work.

### 2.0 MILLTOWN DAM SITE

The 8.5 m high Milltown Dam construction was completed in 1907 at the confluence of the Clark Fork and Blackfoot Rivers (Figure 1). Mean annual flows for the Clark Fork River below the dam (USGS station # 12340500) ranged from 70 m$^3$/day to 86 m$^3$/day during the 2006 to 2010 study period. Over the last century, the Milltown reservoir filled with mining and smelter wastes from the Butte and Anaconda area located 180 km upstream. The Milltown Reservoir was designated a CERCLA (U.S. EPA Superfund) site in 1983 as water seeping from the reservoir sediments recharged the adjacent coarse-grained aquifer and contaminated local wells with dissolved arsenic, iron, and manganese ((ARCO) Atlantic Richfield Company, 1992; Harding Lawson Associates (HLA), 1987; Moore and Woessner, 2002; Udaloy, 1988; Woessner et al., 1984).
The river systems and reservoir are located in a semi-confined mountain valley setting in which the valley floor sediments (predominantly 6 to 60 m thick) are dominated by fluvial sand, gravel, cobbles, and boulders deposited by the ancestral Clark Fork and Blackfoot Rivers. Additional coarse grained sediments were deposited during the Glacial Lake Missoula floods. These sediments are bounded by steep mountain boundaries composed of argillite, quartzite, and limestone metasediments of the Precambrian Belt Series (Gestring, 1994). Residents located adjacent to and proximal to the reservoir utilize the prolific unconfined valley aquifer (with water table depths of 2 to 35 m below land surface) for all domestic and municipal water supplies (Berthelote and Woessner, 2009).

In 2004 the U.S. EPA, the State of Montana, and stakeholders decided to remove the 8.5 m high Milltown Dam and 1.9 of the 5.0 mcm (million cubic meters) of contaminated reservoir sediments to restore groundwater quality, provide fish passage, and return the two rivers to a natural free-flowing state (River Design Group et al., 2008; Westwater Consultants et al., 2005). The Milltown Dam and the associated reservoir were removed during the period of 2006 to 2009. Remediation and restoration plans were designed to be completed in stages over a number of years (Envirocon, 2006; River Design Group et al., 2008). This estimated $100+ million remediation/restoration project required three planned drawdowns starting in 2006 (3.6 m in June 2006, 3.6 m in March 2008, and 1.3 m in April 2009) that correlated with engineering tasks prior to reaching the final free flowing state in 2009.

Before the dam removal process formally began, project engineers initiated a 3.5 m temporary drawdown in November of 2005 to examine the submerged portion of the dam. It was observed that groundwater levels in some wells adjacent to the reservoir declined and a few shallow domestic wells became inoperable. These observations prompted the development of an assessment tool that could be used to identify wells that would likely be impaired and preemptively replace or remediate them prior to the initiation of the next phase of drawdowns. As the second and third (final dam removal) drawdowns occurred, the assessment tool was revised to incorporate new observations and to evaluate if additional remediation was needed.
3.0 CONCEPTUAL MODEL OF IMPACTS TO GROUNDWATER LEVELS FROM DAM AND RESERVOIR REMOVAL

Relying on the literature and basic hydrogeological theory, Berthelote (2013) developed generic conceptual models of how groundwater systems associated with dam removals would respond under various hydrogeological and geomorphic settings. At the Milltown site, groundwater conditions were generally higher than river levels in the river section above the dam and lower than river levels at and below the reservoir area. Berthelote’s (2013) conceptual models fit to Milltown conditions would combine semi-confined high water table (above the dam) and low water table (below the dam) conceptualizations (Figure 2). This model illustrates the progression of hydrogeologic changes expected in Milltown following the breach of the dam. Above the dam, in the high water table setting, slight alterations in the surface water groundwater exchanges will ensue in response to removal of the reservoir and river reconstruction/restoration activities (new gradients, river configurations, bank stabilization, etc). Proximal to and below the dam it is anticipated that vertical channel bed gradients (influent conditions) will all convert to or remain downward, and valley wide groundwater levels will decline with the magnitude of changes being inversely proportional to the distance down gradient from the dam.

![Figure 2](image)

**Figure 2** Hybridization of Berthelote’s (2013) conceptual models fit to Milltown conditions which combines his semi-confined high water table (above the dam) and low water table (below the dam) conceptualizations. This model illustrates the progression of hydrogeologic changes expected in Milltown following the breach of the dam. Generalized hydrographs for wells above and below the dam are presented to demonstrate likely groundwater responses to dam removals immediately following a breach and after the hydrogeologic system is fully restored.

Prior to implementing any monitoring, modeling, or mitigation strategies, project managers understood that they would need to employ adaptive mitigation management strategy that relied on historical data and interpretations, newly collected groundwater level and flux data, and professional knowledge. It was understood from the initiation of the project that potential alterations to the planned engineering activities (timing of each staged drawdown, magnitude of the individual drawdowns, timing of the diversion into a planned bypass channel), issues with monitoring data (sparse or erratic historical data and maintaining consistent access to monitoring or private wells), multiple modeling calibration issues (irregular or transient data sets that would be updated as new data became available), and other transient hydrological concerns would likely arise throughout the project. The key goal of this work was to provide project managers (regulators) with the adequate tools and knowledge that they could use to plan and implement water supply remediation. The timeline in Figure 3 illustrates the accessible transient knowledge base. The subsequent methods section describes the key methodological components of this timeline. Though the project extended over 5+ years, the first wells were mitigated in April of 2006, only one month after the monitoring and modeling process commenced. The time constraints for all mitigation actions (majority
completed during 2007 and early 2008) dictated that the first year of data collection and analyses would formulate the key information used to make the majority of mitigation decisions.

![Project Timeline](image)

**Figure 3** A timeline summarizing the sequencing of events described in this work (vertical columns), changes in site conditions, groundwater level data collection, and the timing and number of wells mitigated in response to modeling results. Data for this figure is located in Appendix 2A

### 4.0 METHODS

To forecast the spatial distribution and magnitude of groundwater level change, a strategy was developed to collect appropriate data sets and to perform required detailed analyses. The strategy had five primary parts; 1) locate and correlate all historical groundwater data; 2) establish a well monitoring program to evaluate groundwater level changes throughout the life of the project including the cataloging of well construction and pumping parameters; 3) construct a three dimensional groundwater model that was capable of adequately representing the historical and current groundwater levels and fluxes; 4) use the model to geospaically and temporally forecast likely groundwater level declines resulting from each staged drawdown and final dam removal; and 5) develop a risk based decision tree to assist project managers with mitigation activities.

This methods section will first present a detailed description of the water level monitoring network and the establishment of background pre reservoir drawdown groundwater level conditions. Next, a brief description of the construction, calibration, and application of an industry standard three dimensional numerical groundwater model is presented. Thirdly, the development of a risk based decision tree and its application to well mitigation is described.
4.1 Baseline Conditions and Groundwater Level Monitoring

4.1.1 Well Monitoring Network
The monitoring network consisted of 78 wells located at 56 locations (Figure 1). Groundwater levels at 22 wells were recorded at intervals no greater than 60 minutes using Solinst® continuous water level recorders (recording pressure transducers corrected with readings from a separate Solinst® barlogger). The remaining wells were measured monthly using an electric water level tape. Appendix 2B contains a compilation of historical water level data and data derived from this work (1981 to 2010).

4.1.2 Surface Water Monitoring Network
A network of surface water stage gauges was established. Surface water elevations were obtained from USGS gauging locations on the Blackfoot River at Bonner (#12340000), Clark Fork River at Turah (#12334550) and Clark Fork River above Missoula (#12340500). Monthly river stage data were supplemented with project installed staff gauges and a continuous water level recorder operated at the Milltown Dam (prior to stage 2 drawdown). Initially, the staff gauge spacing was set up at approximately equal distant intervals (no greater than every 800 meters) along the river channels depending on river accessibility (Figure 1). Both river stage gauges and wells without established measuring point elevations were surveyed using a real-time kinematic survey-grade Trimble 5800 GPS surveyor using standard techniques (Trimble, 2008).

4.1.3 Water Level Data Analyses and Establishment of Baseline Conditions
Water level data were analyzed by constructing and evaluating well hydrographs, flow nets, and regional water table maps, and reviewing river discharge and climatic data. Spatial and temporal water level data were used to calibrate numerical models and assess the net change in groundwater levels from simulated and observed changes during dam removal.

A second phase of the analyses identified sets of water level data that could be used to compare groundwater conditions prior to dam removal with forecast and observed water levels following dam removal. A database was compiled with groundwater data from 1982 to 2006. It was stratagized that an effort should be made to forecast the maximum likely reduction in the water table position so remediation decisions would minimize the need for additional future work. As no pre-reservoir construction groundwater data were available, this involved establishing a pre-dam baseline data set to be combined with model forecasts of changes. Baseline development required a sufficient spatial and temporal groundwater level data set that represents the area likely to be impacted, and information of the natural variation of seasonal water levels. Previous groundwater studies found the lowest groundwater levels occurred in late winter, a period when river flows and river stages (Gestring 1994). Also, previous work had shown annual groundwater levels were lower than annual monthly averages during periods of less than normal stream flow (Berthelote 2013). The most spatially complete data set was derived as part of this study (2006-2007). The lowest groundwater level conditions were associated with March 2006. In addition, low stream flows conditions had occurred in the previous year (64% probability of exceedence). Based on data availability, March 2006 water table elevation data were used to represent the pre dam removal base case from which modeling results would be combined to provide forecasts of post dam removal groundwater conditions.

4.2 Data Sets Needed to Formulate a Predictive Groundwater Model
Interpretation of the groundwater level data and stream stage data sets as well as review of previous investigations related to the Milltown CERCLA site supported the reported aquifer conditions that allowed for the transmission of large volumes of groundwater. A water budget was prepared for the study site.

4.2.1 Water Budget
An annual groundwater balance for the study area was formulated as follows:

\[
\text{In} = \text{Out} +/- \text{Change in Storage} \\
GWinCFR + GWinBFR + GWinDC + GWinMC + BFleak + CFRleak + Resleak + GWinBR = \\
GWoutCFR + GWConsP + GSWout +/- GWS
\]
where:

GWinCFR is lateral groundwater underflow into the model area from the Clark Fork River Valley at Turah Bridge; GWinBFR is lateral groundwater underflow into the model area from the Blackfoot River valley; GWinDC is lateral groundwater underflow into the model area from Deer Creek; GWinMC is lateral groundwater underflow into the model area from Marshall Creek; BFRleak is seepage (recharge) from the Blackfoot River channel into the valley aquifer; CFRleak is seepage (recharge) from the Clark Fork River channel into the valley aquifer; Resleak is seepage (recharge) from Milltown Reservoir into the underlying aquifer; GWinBR is the seepage into the model domain from a bedrock groundwater system; GWoutCFR is lateral groundwater underflow from the Clark Fork River Valley at Hellgate Canyon out of the model domain; GWConsP is consumed groundwater pumped from wells; GSWout is groundwater seepage into the Clark Fork River within the model area; +/- GWS is the net change in groundwater storage (volume of water annually removed or added to the aquifer) (Figure 4).

Figure 4 Generalized conceptual model illustrating the components of the water balance.

The groundwater flow system was strongly influenced by Clark Fork River and Blackfoot River leakage at the reservoir site and downstream. A number of the water budget components were estimated using Darcy Law calculations based on aquifer thickness estimates, local gradients, and estimated hydraulic conductivities. Initial aquifer property information was compiled from existing studies (ARCO Atlantic Richfield Company, 1995; Newman, 1996; Woessner et al., 1984), study collected data (Table 1) (Berthelote et al., 2007), and previous modeling studies (Brick, 2003; Gestring, 1994). Hydraulic conductivity data sets were extrapolated by analyzing well hydrograph data using stage peak lag time methods (Pinder et al., 1969), flow net analyses (Fetter, 2001), and historical pumping tests (Harding Lawson Associates (HLA), 1987; Walton, 1987; Woessner and Popoff, 1982).
Table 1: Table of techniques, methods, and equipment used to generate data sets describing the hydrological system in the study area.

<table>
<thead>
<tr>
<th>Technique</th>
<th>Method / Equipment</th>
<th>Error</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Single Point Survey</td>
<td>Trimble XPS Survey Grade GPS</td>
<td>0.02 m</td>
<td>(Trimble, 2008)</td>
</tr>
<tr>
<td>River Stage</td>
<td>USGS Gage</td>
<td>0.003 m</td>
<td>(USGS, 2011)</td>
</tr>
<tr>
<td>Study Installed Staff Gage</td>
<td>0.003 m</td>
<td>(Berdhule et al., 2007; Geering, 1994; Tallman, 2005)</td>
<td></td>
</tr>
<tr>
<td>Study Installed Continuous Data Recorder</td>
<td>0.07 m</td>
<td>(Berdhule et al., 2007; Geering, 1994; Tallman, 2005)</td>
<td></td>
</tr>
<tr>
<td>Groundwater Levels</td>
<td>Steel or Electronic Tape</td>
<td>0.03 m</td>
<td>(Berdhule et al., 2007; Geering, 1994; Tallman, 2005)</td>
</tr>
<tr>
<td>Vertical Hydraulic Gradients</td>
<td>Potentiometer</td>
<td>0.005 m</td>
<td>(Caldwell and Hawes, 2003; Tallman, 2005)</td>
</tr>
<tr>
<td>V/SDRE</td>
<td>0.005 m</td>
<td>(Hussch et al., 2000; Johnson et al., 2005)</td>
<td></td>
</tr>
<tr>
<td>Thermal Remote Sensing</td>
<td>Qualitative</td>
<td></td>
<td>(Loeng et al., 2005; Stanford et al., 2005)</td>
</tr>
<tr>
<td>Stream Bed Conductivity</td>
<td>Falling Head Test</td>
<td>3%</td>
<td>(Landon et al., 2002)</td>
</tr>
<tr>
<td>Aquifer Conductivity</td>
<td>Peak Lag Analysis</td>
<td>6%</td>
<td>(Pinder et al., 1969)</td>
</tr>
<tr>
<td>Porosity &amp; Specific Yield</td>
<td>Historical Pumping Tests</td>
<td>6.4%</td>
<td>(Halen, 1966; Land and Water Consulting, 2005; Walton, 1987)</td>
</tr>
<tr>
<td>Storativity</td>
<td>Literature</td>
<td>-%</td>
<td>(Fetter, 2001)</td>
</tr>
<tr>
<td>Bedrock-Aluvium Boundary</td>
<td>Well Logs</td>
<td>1.5 m</td>
<td>(UCW, 2008)</td>
</tr>
<tr>
<td></td>
<td>Slope Projections</td>
<td>- m</td>
<td>(Krock, 2003; Janiszewski, 2007; Nyquest, 2001)</td>
</tr>
<tr>
<td></td>
<td>Base Holes</td>
<td>0.6 m</td>
<td>(Janiszewski, 2007; Nyquest, 2001)</td>
</tr>
<tr>
<td></td>
<td>Seismic Lines</td>
<td>4.5 m</td>
<td>(Gradient Geophysics, 1991; Janiszewski, 2007)</td>
</tr>
</tbody>
</table>

Though many wells derive water from the aquifer, most wells do not extend through the full saturated thickness of the aquifer and into the underlying bedrock. The geometry of the lateral and bottom boundaries of the unconsolidated valley aquifer were revised by compiling and analyzing historical borehole data, construction site borings, well logs, topographic projections of mountain slopes into the subsurface as well as some previous surface geophysical estimates of bedrock depths (Figure 5) (Evans, 1998; Gradient Geophysics, 1991; Nyquest, 2001; Sheriff and others, 2007; Woesnner et al., 1984). To further refine aquifer saturated thickness estimates, additional gravity data were collected and interpreted using a Scintrex CG3 Micorgal Gravity Meter.

Figure 5: Spatial distribution of geophysical data, and site boring and well log data used to estimate bedrock depths.

In order to establish spatial and temporal surface water and groundwater exchange rates and locations before and after dam removal, several direct in-channel measurement techniques were applied (Figure 6). This work established the locations, directions, and rates of river leakage into the aquifer and groundwater discharge to the river channel. The assessment also compared river stage elevations to nearby shallow well water levels and interpreted water table maps to evaluate if a section of river channel could be classified as gaining, losing, flow-through, or parallel flow (Woesnner, 2000).
Using river stage, VHG (vertical hydraulic gradient) and flux calculations, and analyses of water table maps, channel reaches were assigned as either losing or gaining. Point data were extrapolated by assuming they were representative of conditions one half the distances between adjacent data collection sites. Quantitative assessment of the exchange rates required installation of instruments to characterize vertical hydraulic gradients, river bed hydraulic conductivities, and river bed flux rates using falling head tests with single or clusters of steel piezometers (Baxter, 1977; Bouwer, 1989; Farinacci, 2009; Sealon et al., 2002). Vertical temperature arrays were also installed in the river bed to estimate flow directions and fluxes (Constantz et al., 2003; Farinacci, 2009; Hsieh et al., 2000; Johnson et al., 2005).

### 4.3 Formulation of the Numerical Groundwater Modeling

More than two decades of hydrologic investigation of the Milltown Reservoir Superfund site resulted in the construction of two earlier two dimensional numerical groundwater models of portions of the study area (Figure 1) (Brick, 2003; Gestring, 1994). For this research, a third set of numerical models (Y1, Y2 and Y3 Figure 3) were developed using Ground Water Vistas graphical user interface to the USGS MODFLOW code (ESI, 2004; Harbaugh, 2005; Harbaugh et al., 2000). These three dimensional groundwater models contained up to 7 layers and were discretized into 44,837 active 46 by 46 m cells. Initial forecasts of water level responses were based on original staged drawdowns and dam removal engineering plans. The response of the groundwater system to stage 1 drawdown was included in the year one (Y1) model forecasts and each of the subsequent drawdowns were incorporated into revised models (Y2 and Y3) which produced new sets of forecasts (Berthelote et al., 2007; Berthelote and Woessner, 2008, 2009; Berthelote et al., 2010).

Boundary condition, hydraulic conductivity and storage coefficient distributions, river reservoir stages, and river bed hydraulic conductivities were principal model inputs. Hydraulic conductivity data were assigned to zones based on previous modeling efforts and field data, knowledge of the sediment distribution, professional judgment, and borehole stratigraphy-hydraulic conductivity relationships (Berthelote et al., 2007; Berthelote and Woessner, 2008, 2009; Berthelote et al., 2010). Over 780,000 input parameters were required to populate the model. The groundwater flow system was simulated in both steady state and transient conditions. Model calibration was achieved by comparing head responses, river-groundwater spatial and temporal exchange rates, and computed lateral down-valley groundwater flows into and out of the study area. Both trial and error and parameter optimization methods (Pest pilot points and regularization) were applied (Anderson and Woessner, 1992a; Berthelote et al., 2007; Berthelote and Woessner, 2008, 2009; Berthelote et al., 2010; Doherty, 2000). During the three years of analyses, the base model (Y1) was first calibrated to a base case (minimum annual groundwater level) steady state conditions (March 31, 2006), transient conditions (March 31, 2006 to April 21, 2007), and history matched with October 8, 1992 steady state data and a transient data set, 10/8/92 to 7/7/93 (e.g. Anderson and Woessner).
The final Y3 model set utilized MODFLOW 2000 (Berthelote et al., 2010; Harbaugh et al., 2000) to forecast and test calibrations using both steady state and transient conditions (1992 to 1993 and 2006 to 2010). Appendix 2C contains the final set of input files. Complete documentation of the model development, calibration, forecasts, and uncertainty analyses are presented in a number of project reports (Berthelote et al., 2007; Berthelote and Woessner, 2008, 2009; Berthelote et al., 2010). A summary table of groundwater and surface water model inputs is found in Berthelote, 2010, Appendix B. The forecast March water levels for individual stages of reservoir drawdown were compared to the March 2006 water level reference in order to compute water level change impacts from planned reservoir drawdowns.

It is clearly recognized that predicting future groundwater responses from planned reservoir stage changes have many challenges. In addition to having data on the planned reservoir stage declines, timing, and levels, assumptions were needed to represent the stream stage and flows, changes in aquifer recharge, and river leakage. These conditions were generally estimated from construction designs, by assuming the river bed properties did not change and the stream stages mirrored the 2006-2007 hydrograph for steady state March water level forecasts. Stream stage data within and immediately proximal to the reservoir were modified during the modeling process to reflect the changes in reservoir stage, physical reservoir channel changes, and stream bed elevations.

4.4 Development of Alternative Conceptual Models /Uncertainty Analyses
Parameters assigned to active model nodes, and hydrological changes in stream stages and lateral valley inflows and outflows are required to calibrate transient and steady state models. However, these models do not produce unique solutions. Though results from a single calibrated model may become the principle tool used to assess impacts from dam removals, a methodology to assess uncertainty in model forecasts is needed. Two approaches to defining prediction uncertainty are often exercised: 1) development of alternative conceptual models (variations in parameters or physical changes in boundaries and/or source of sink terms that bracket likely ranges of predicted impacts); or 2) geostatistical model averaging or analyses (random variation of key parameters constrained by assigned probability density functions) (Singh et al., 2010; Ye et al., 2010). Our approach was to conduct a standard sensitivity analyses on the extensively calibrated model used to make post dam removal groundwater level forecasts (Anderson and Woessner, 1992a; Hill, 1998; Hill et al., 1998; Hill and Tiedeman, 2007), and then develop alternative parameterization that would likely bound (create a range of key sensitive values) predictions.

The sensitivity analyses uniformly varied each zoned group of parameters (formational hydraulic conductivity, river bed hydraulic conductivity, specific yield, and storativity,) by +/- 10 and 20% while holding all other parameters at the calibrated model value (Anderson and Woessner 1992). Changes in the Root Mean Squared Error (RMSE) of the difference between observed and simulated heads were used to assess the sensitivity of each parameter to the assigned changes. Parameters were considered sensitive if any zone variation of 10 or 20% resulted in a RMSE change greater than or equal to 0.1m. Once the sensitive parameters were identified a combination of the most sensitive parameters was used to create a set of alternative models to bracket the calibrated model forecasts. The two most sensitive parameters (horizontal hydraulic conductivity and river bed conductance) were chosen to develop the first two alternative conceptual models. One alternative model used a combination of uniformly raising the hydraulic conductivity values by 20% and uniformly lowering the river bed conductance by 20% to produce the likely lowest forecast water table. The second alternative model reversed the magnitude change of these parameters to produce the likely highest forecast water table. These models were used to evaluate how reasonable combinations of sensitive parameters changed model forecasts of reservoir drawdown impacts.

A second set of alternative conceptual models tested the impact of uniformly changing the elevation of the bottom boundary condition +/- 5 m (an uncertainty value identified in the geophysical data analyses). Decision tree analyses using the original forecast model results and the alternative model results provided managers with a range of the number of wells likely to require some form of remediation.

4.5 Mitigation Process

4.5.1 Model Forecasting and Mitigation Requirements
Forecast post drawdown groundwater level minimums (March) were used to develop a decision tree/risk analyses methodology (Figure 7). Threats of reducing or losing productivity from individual wells were
ranked. The wells closest to the reservoir or having a shallow depth were investigated first. The database we developed using standard well drillers logs (online Montana Bureau of Mines and Geology GWIC) contained well depths but no information on the depth of the pump intake. As not all wells had well logs and in most cases pump setting depths were absent, a large number of wells were field visited and additional data collected. These data were then assessed using a decision tree developed in consultation with the regulators, project managers and stakeholders (Table 2).

Figure 7  A) Outline of research objectives for each year; B) Work flow required to identify the level of risk of impacting groundwater supplies and the steps used to select wells for mitigation.

Decisions fell into five categories (Table 2). No action (OK) was recommended when a forecast water level was at least one meter above the elevation of the top of the pump, and the bottom of the well (or well intake) was greater than three meters below the predicted water level. Second, the well was replaced (NEW) because the forecast water level was less than three meters above the well bottom. Third, lower the pump (LOWER) because more than three meters of water were in the well bore but the pump set results in less than one meter of water over the pump. Fourth, Check the pump set (CHECK) as the forecast water level was greater than three meters but the pump set elevation is unknown. Fifth, Pull the pump (PULL) to determine pump set or well depth because they were unknown and can't not be determined by sounding the well. Table 3 outlines the decision paths that were used to make specific recommendations.

Table 2  General mitigation decision path for seven hypothetical well construction scenarios. Well scenario numbers and elevations (in meters) are shown strictly as examples and do not relate to well numbers at the study site. The predicted water elevation (water table) (B) is representative of the model forecast results used for each analyses (calibrated or alternative conceptual model prediction).

<table>
<thead>
<tr>
<th>Well Scenario</th>
<th>A</th>
<th>B</th>
<th>C</th>
<th>D</th>
<th>Decision</th>
<th>Action</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 920 914 917 915</td>
<td>If B &lt; D + 2</td>
<td>New</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2 920 914 912 908</td>
<td>If B &gt; D + 3 &amp; C &lt; B - 1</td>
<td>OK</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>3 920 914 917 908</td>
<td>If B &gt; D + 3 &amp; C &gt; B - 1</td>
<td>Lower</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>4 920 914 ? 908</td>
<td>If B &gt; D + 3 &amp; C is unknown</td>
<td>OK</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5 920 914 ? 913</td>
<td>If B &lt; D + 3 &amp; C is unknown</td>
<td>Check</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>6 920 914 912 ?</td>
<td>If B &gt; C + 1 &amp; D is unknown</td>
<td>OK</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>7 920 914 917 ?</td>
<td>If B &lt; C + 1 &amp; D is unknown</td>
<td>Pull</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

5.0  RESULTS
The dam removal process at Milltown was not a single operation taking only a few days or weeks to complete. Because the reservoir was filled with contaminated sediments, the dam and reservoir removal plan was staged. The process included removal of 1.9 of the 5 mcw (million cubic meters) of sediment which took three years. The sediments removal action was designed to lower the reservoir stage to allow
dewatering of portions of the reservoir sediments so they could be “dry” excavated. In addition, annual
drawdowns were planned to minimize river metal, arsenic and turbidity conditions. Initially the stage 1
drawdown was initiated on June 1st, 2006 after the spring river discharge peak. Due to surface water quality
concerns, it was suspended from July 7th, 2006 to September 18th, 2006 and then resumed. Final Stage 1
drawdown was completed on November 12th, 2006. The maximum stage 1 drawdown from full pool was
~3.6 m (Figure 3). To avoid possible future summer water quality issues Stage 2 drawdown (additional 3.6
m) began on March 28, 2008 and the final Stage 3 (additional 1.3 or 8.5 m total) drawdown was completed
on March 27, 2009 with the removal of the spillway coffer dam.

5.1 Groundwater Base Case
Establishment of a base case from which to reference observed and forecast impacts was complicated at
this site by the availability of only short, intermittent, and spatially discontinuous historical groundwater
level data. To overcome this limitation, all water level data at each historical monitoring well were plotted
by month of collection on a 1 year time scale. (1982 to 2006) (Figure 8). An envelope surrounding the
points was used to indicate temporal variability (gray shading). It was decided that initial data collected as
part of this work, water levels for March 2006, would be used to generally represent the pre dam removal
groundwater conditions from which the groundwater impacts would be measured. The March 2006 data
were consistently near the historical lower boundary of the groundwater level position shown in the
historical measurements. A plot of project collected 2006 to 2007 water table data shows a response to the
June through September Stage 1 reservoir decline. Groundwater level impacts resulting from reservoir
drawdowns and removal were generated by using numerical models each of the first three project years
(Y1, Y2 and Y3) to forecast future March groundwater conditions and then subtracting these elevations
from the March 2006 base case values.

![Figure 8](image-url) Hydrograph of Well 01 (closest well to the dam) showing the relative variation of 1982 to 2006 historical
groundwater level elevations (squares and gray shaded area) and 2006 to 2007 groundwater levels with respect to a
March 2006 groundwater level (used as the reference point from which to measure change) (see Appendix 2B for
complete data sets).

5.2 Observed Groundwater Changes during Stage Drawdowns and Complete Dam removal
Water level data collected between 2/24/1982 to 5/9/2010 included 1,805 groundwater monitoring days
with over 87,245 measurements at 229 locations and 1,276 surface water monitoring days with over 5,028
measurements at 25 locations (Appendix 2B). March water table maps for each year (1993, 2006, 2007,
2008, 2009, & 2010), and a representative hydrograph from a well near the dam (well 05) show that
following the dam and reservoir removal the general flow directions and seasonal water level responses
(2010) remained similar to pre drawdown conditions (1993 and 2006) (Figure 9). The difference between
the base case observed water levels and the observed 2010 water levels are illustrated in Figure 10A. In
addition, observed March groundwater levels for each subsequent year were differenced from the base case
March 31 2006 groundwater conditions to quantify observed changes. Though these data show changes in
groundwater levels, deciphering the influence of the magnitude of changes resulting solely from the
drawdowns and dam removal is complicated and can be highly dependent on previous streamflow
conditions and recharge regimes (Berthelote, 2013 Chapter 3). During the dam removal period both above
and below average stream flow conditions occurred (Figure 9). These conditions likely resulted in variable
rates and durations of surface water groundwater exchanges that directly influenced overall observed groundwater level elevations. When dam removal groundwater impacts are based on base case and observed post-dam removal measurements, portions of the groundwater system declined 2 to 3 m as a consequence of the reservoir and dam remediation activities. Mapped water level declines were not limited to the immediate reservoir area but rather extended at least 6 km downstream and 2 to 3 km upstream (Figure 9).

**Figure 9** Observed groundwater levels. Valley water table maps for March low water levels from 1993, and 2006 to 2010. Water level contours indicate that groundwater flow directions remained uniform in areas above and below the dam site, however some variations were noted proximal to and north of the reservoir area (central portion of the map). The bottom graph illustrates the transient groundwater hydrograph for a well immediately below the dam site (well 05). River discharge is presented for the above Missoula USGS gauge for comparison. The red dots highlight the March 31 water level for each of the project years.
5.3 Water Budget and Surface Water-Groundwater Exchange

Initial pre-dam removal field data revealed the rivers and groundwater systems formed a complex hydrogeological system. Groundwater recharge is primarily from four sources: 1) a perched and leaking (losing) Blackfoot River and Clark Fork River arms of the reservoir and most of the Clark Fork River from just below the dam to Hellgate Canyon (vertical riverbed hydraulic conductivities range from 0.4 to 12.8 m/d and leakage rates from 68,000 to 284,000 m³/d); 2) lateral down-floodplain underflow at the Turah Bridge Boundary in (11,900 m³/d to 45,300 m³/d), 3) underflow from the Blackfoot River floodplain into the Clark Fork valley (1,600 m³/d to 19,500 m³/d) and, 4) limited recharge from the mountain boundaries (less 4400 m³/d) (Berthelote et al., 2007; Farinacci, 2009; Tallman, 2005; Woessner and Popoff, 1982).

Aquifer discharge occurred in a small gaining reach directly below the dam (300 m), as underflow through the valley aquifer located in Hellgate Canyon to the west (59,400 m³/day to 566,300 m³/day), and locally in gaining stream sections of the Clark Fork River above the reservoir. Recharge from precipitation and discharge from evapotranspiration were considered to be negligible as they were estimated to be a small percentage of the total water balance (Woessner et al., 1984). Groundwater inflow from the bedrock boundary was also estimated to be small and not a significant component of the water balance (Woessner et al., 1984). The groundwater was assumed to be at steady state for the year period.

Generally, groundwater flowed towards the reservoir area from the upper Clark Fork River valley. This component of the flow system converged just north of the reservoir with groundwater moving from the Blackfoot River Canyon south towards the reservoir area (Figure 9). These systems combined with the northward moving groundwater from the reservoir and the northwesterly groundwater flow continued west eventually discharging as underflow through Hellgate Canyon.

The aquifer is highly conductive (range of hydraulic conductivity from 90 to >27,000 m/d with a mean of 3600 m/d) with groundwater velocities calculated in tens to hundreds of meters per day. Hydraulic gradients range from 0.0015 (upper Blackfoot River arm) to 0.066 (near the dam), and estimated porosities are 0.20 (Berthelote et al., 2007; Berthelote et al., 2010; Gestring, 1994; Moore and Woessner, 2002; Tallman, 2005; Woessner et al., 1984). Surface water and groundwater exchange locations and volumes ranged from 2.3 to 43 m³/(day m) as computed using hydraulic properties, gradients, and temperature modeling (Table 3).

### Table 3 Water Balance Summary: Groundwater inflow outflow spatial descriptions, magnitudes, ranges, potential errors, and sources

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Minimum m³/day</th>
<th>Maximum m³/day</th>
<th>Error</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>GW under Hellgate Canyon</td>
<td>9.0E+01</td>
<td>5.1E+04</td>
<td>±2.2E+04</td>
<td>Tallman A.A., 2005</td>
</tr>
<tr>
<td>GW under Blackfoot River</td>
<td>3.96E+00</td>
<td>6.8E+02</td>
<td>±2.2E+02</td>
<td>Tallman A.A., 2005</td>
</tr>
<tr>
<td>GW under Clark Fork River</td>
<td>3.96E+00</td>
<td>8.8E+02</td>
<td>±2.2E+02</td>
<td>Tallman A.A., 2005</td>
</tr>
<tr>
<td>GW under Clark Fork River</td>
<td>3.96E+00</td>
<td>1.1E+06</td>
<td>±2.2E+06</td>
<td>Tallman A.A., 2005</td>
</tr>
</tbody>
</table>

5.4 Numerical Modeling Results

5.4.1 The forecasting of groundwater

Forecasts of future groundwater levels were generated with extensively calibrated models. Simulated groundwater levels were calibrated so they were consistently less than 2 m different from observed values (up to 77 network wells depending on the model run). The resulting model water budgets were within the targeted baseline pre drawdown ranges of uncertainties presented in Table 3 (Table 4). Fitted hydraulic conductivity and river leakage parameters were within measured ranges. The simulated water balance...
results were consistently stable and differences between inflow and outflow components were less than 0.02 percent.

The simulated Y1 base case steady state water budget compared favorably with the pre-model estimated steady state budget (Table 4). Seepage from the Clark Fork River into the underlying groundwater and the flow of valley groundwater into gaining portions of the river were slightly less than original budget estimates. The calibrated steady state heads, boundaries, and fluxes were used as initial conditions during transient model calibration. The comparison of simulated heads to observed heads (March 31, 2006 to April 21, 2007), and the pre-modeling estimated water balance revealed the model reasonably produced observed conditions under this more demanding evaluation. For the transient simulation 60 observed heads were used as calibration targets. They were distributed as follows: 23 in layer 1; 7 in layer 2; 8 in Layer 3; 7 in layer 4; 8 in layer 5; 3 in layer 6, and 4 in layer 7. Results comparing observed and simulated heads over time show relatively good fits of simulated water level positions with observed levels at most sites.

Table 4  Comparison of the pre-model estimated (includes error estimate) and simulated steady and transient state water balances. Transient water balance was presented for the last stress period of April 21, 2007.

<table>
<thead>
<tr>
<th></th>
<th>Estimated range (m³/day)</th>
<th>Y1 Modeled value (SS 3/31/06) (m³/day)</th>
<th>Y1 Modeled value (T 4/21/07) (m³/day)</th>
</tr>
</thead>
<tbody>
<tr>
<td>GW in MC</td>
<td>1.2* 10⁻⁴ - 4.6* 10⁻⁴</td>
<td>3.7* 10⁴</td>
<td>3.4* 10⁴</td>
</tr>
<tr>
<td>GW in DC</td>
<td>4.6* 10⁻⁴ - 2.4* 10⁻⁴</td>
<td>1.4* 10⁴</td>
<td>5.1* 10⁴</td>
</tr>
<tr>
<td>GW in CF</td>
<td>1.6* 10⁻⁴ - 5.1* 10⁻⁴</td>
<td>2.5* 10⁴</td>
<td>2.5* 10⁴</td>
</tr>
<tr>
<td>GW in MC</td>
<td>3.1* 10⁻⁴ - 4.6* 10⁻⁴</td>
<td>8.8* 10⁴</td>
<td>8.8* 10⁴</td>
</tr>
<tr>
<td>BFR in</td>
<td>1.4* 10⁻⁴ - 9.9* 10⁻⁴</td>
<td>5.7* 10⁴</td>
<td>4.2* 10⁴</td>
</tr>
<tr>
<td>CFR in</td>
<td>3.1* 10⁻⁴ - 5.1* 10⁻⁴</td>
<td>9.1* 10⁴</td>
<td>1.2* 10⁵</td>
</tr>
<tr>
<td>GW out</td>
<td>2.4* 10⁻⁴ - 1.1* 10⁻³</td>
<td>5.4* 10⁴</td>
<td>2.4* 10⁵</td>
</tr>
<tr>
<td>Total Inflow</td>
<td>1.6* 10⁻⁴ - 1.2* 10⁻³</td>
<td>2.5* 10⁵</td>
<td>2.4* 10⁵</td>
</tr>
<tr>
<td>GW out CFR</td>
<td>6.0* 10⁻⁵ - 5.7* 10⁻⁵</td>
<td>2.3* 10⁵</td>
<td>2.0* 10⁵</td>
</tr>
<tr>
<td>GW out BFR</td>
<td>2.6* 10⁻⁵ - 5.1* 10⁻⁵</td>
<td>2.0* 10⁵</td>
<td>2.4* 10⁵</td>
</tr>
<tr>
<td>Total Outflow</td>
<td>2.6* 10⁻⁵ - 6.6* 10⁻⁴</td>
<td>2.5* 10⁵</td>
<td>4.8* 10⁵</td>
</tr>
</tbody>
</table>

These calibrated models (Y1, Y2, and Y3) were used principally to forecast the March groundwater levels after each stage of reservoir drawdown was completed. Model Y1 and the two hydraulic conductivity and river leakage alternative model forecasts (Y1 Hi and Y1 Low) were used to forecast the response of the system to the stage 2 drawdown. The following year Model Y2 was recalibrated to the initial stage 1 and stage 2 groundwater level response and was used to forecast impacts from stage 3 drawdowns. Finally, model Y3 was revised using all the observed stage-drawdown responses and to forecast the final dam out impacts (Figure 10). To illustrate the type and analyses of model results, water level data for well 01 is presented (Figure 10). The forecast responses at well 01 are indicative of the simulation results at the other 77 wells used for calibration. Transient calibrations of each model had difficulty closely matching groundwater levels at the peak hydrograph periods (Figure 10 C). However, the models (Y1, Y2, and Y3) were well calibrated with all root mean square errors less than 0.51. The alternative conceptual models that employed varying bedrock elevations (Y2 B+15 (ft) and Y2 B-15 (ft)) revealed that changes in this boundary did not extend the range of impacts forecast by the other conceptual models during Y2 modeling (Figure 10B). As a result they were not used as viable alternative conceptual models in Y3 modeling. Examining the results of the calibrated models and alternative models at well 01 suggest that the initial Y1 modeling adequately represented likely groundwater level changes resulting from stage 2 drawdowns. Y2 and Y3 modeling were also reasonable representations of hydrogeologic conditions and useful in forecasting the groundwater response to stage 3 and final dam out actions.
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Figure 10. A) A spatial representation of the difference between the observed 3/31/06 base case and the observed 2010 March groundwater levels. B) Composite hydrograph of well 01 showing a graphical representation of the range of modeling results for the calibrated (green data) and four alternative conceptual models (red, blue, purple, & orange) for each year. The simulation in which the hydraulic conductivity was uniformly lowered by 20% and the river leakage uniformly raised by 20% always resulted in the modeled water table being at a higher elevation (blue points). The simulation in which the hydraulic conductivity was uniformly raised by 20% and the river leakage uniformly lowered by 20% always resulted in the modeled water table being at a lower elevation (red points). Observed groundwater level data are represented by the brown line. The brown horizontal line represents the March 31st 2006 base case pre-removal low water table position. C) A transient illustration of the Y3 modeling results bounded by the alternative conceptual modeling results compared to the observed groundwater levels for well 01.

5.5 Mitigation Process
The goal of the agencies responsible for groundwater impact mitigation was to provide dependable supplies of water to every resident without significant service gaps. Year 1 groundwater modeling results were used by project managers prior to the winter of 2007 to identify groundwater supply wells that were likely to require mitigation (Figure 11A). By the completion of our evaluation (Spring 2008), all 515 surveyed wells were evaluated for possible impacts with 286+ well sites physically visited. Sixty wells were identified and remained in the “check” category as needing additional information. Wells identified as likely to be impacted were mitigated based on the Table 3 decision tree.
Y1 forecasts resulted in identifying between 62 and 137 wells needing some sort of mitigation which included between 42 and 84 wells needing replacement (New), 13 to 36 pumps that would require lowering (Lower) and up to 17 pumps that would need to be pulled (Pull) to determine total well depth. Hundreds of wells were initially identified as needing more information (Check). Based on Y1 results, the field campaign data were used to update the mitigation recommendations. The final results illustrated in Figures 11 and 12 indicate the final numbers generated by the decision tree at the end of the project. As the mitigation analyses proceeded we performed additional modeling refinements; however, project managers continued to base all mitigation decisions on the Y1 modeling scenarios primarily due to the fact that subsequent models were produced after the majority of the wells had been mitigated (Figure 3). Figure 11B illustrates the variance between the actual mitigation actions completed and the recommendations computed by the decision tree. All subsequent modeling efforts confirmed initial actions and only minimally refined the recommended actions (Figure 12).

Using the results of this work, the project manager would inform the well owner of the perceived risk and offer to mitigate the well. Many well owners with well depths recommended for mitigation that were close to the decision tree threshold decided to opt out of immediate mitigation to wait and see what would happen. This was presumably due to the decision makers’ public statement that they would mitigate any...
wells affected by dam removal for up to one year following the final stage 3 removal. Many homeowners whose wells were close to the decision tree threshold, but not recommended for remediation, insisted on a replacement. Often these well owners received new wells. Appendix 2D contains a compilation of the well survey data by location.

6.0 DISCUSSION AND CONCLUSIONS
Groundwater supply mitigation implemented in response to the dam and reservoir removal cost close to $1,000,000. This work showed a well calibrated numerical groundwater model as an appropriate tool with which to forecast groundwater response to the removal action. However, though challenges remain, the applied methodology appears to present a reasonable approach when attempting to identify groundwater impacts.

6.1 Challenges of Predicting Groundwater Impacts
Throughout the mitigation process some wells became inoperable and were mitigated before recommendations were completed. All of these wells were initially identified by the decision tree as needing more information (Check or Pull), and at the time of production loss had not been visited. Several wells were mitigated without the direct need (identified as OK) in an attempt to appease social concerns where small (<1m) differences in the initial forecast would have altered the recommended action from OK to Action. A few of these individual well owners found that the deeper waters had a different chemistry that required the installation and maintenance of treatment systems to remove increased concentrations of manganese, iron, and other minerals.

When the year 1 and 2 observational data was included in the Y3 modeling efforts, it suggested that the earlier modeling had over estimated water table changes. This is attributed to an underestimation of the hydrogeologic properties assigned to represent the bed of a temporary bypass channel constructed to assist in removal of some portions of the reservoir sediments. Further model calibration revealed channel leakage into the groundwater from the bypass channel needed to be doubled to meet calibration targets. The increased leakage acted to reduce forecast groundwater level declines and bring them more inline with observations. Attempts were made to verify these calibrated rates by instrumenting the new channel, however high river flows resulted in the loss of field instrumentation, and access was limited by construction activities.

Though the selection of the base case data set (March 2006) was dictated by available historical data and was within a low stream discharge year (64% probability of exceedence), it is not unlikely that future groundwater levels will be less than those predicted by the model (36% probability of lower river discharge years or drought). Efforts should be made to forecast future minimal flow groundwater levels to make this impact analyses more complete. This raises the question of what happens in the future if water level decline in wells so that water supplies are interrupted? Certainly mitigation decisions and actions that formed this work should be well documented in anticipation of further questions that may arise regarding quantifying impacts from removal actions. In this setting, the river exchange with the groundwater controls the seasonal variations and overall position of the water table, thus the maximum and minimum annual water table elevations. During low flow drought periods water table positions are lower than average and wetter than normal periods have the opposite affect. It is recognized that this 2006 to 2010 climate signal is over printed on the observed water table response to dam removal. How the river stage and flows will behave in the forecast period is of course unknown, thus the predicted response of the groundwater system does not mirror 2010 observations. This makes testing the model forecast difficult as pre established variations in groundwater conditions were assumed to produce the forecast. In the Milltown case, only if the antecedent flow and groundwater conditions matched the March 2006 water table elevations could the accuracy of the forecast be evaluated.

6.2 Evaluation of Remediation Actions
In most cases, the proactive mitigation activities were completed prior to a well becoming inoperable, so any direct measure between the recommended number of wells needing action and the number actually needing mitigation is difficult. This is because old wells were immediately sealed when the new ones became operational. Thus, a direct comparison of the mitigation vs. recommendation numbers (as presented in Figures 11 and 12) does not provide an evaluation of the effectiveness of the mitigation process. The
new well can, however, provide a proxy for the resultant groundwater elevation due to the close proximity of the two wells. If, however, actual observational data (reactive) rather than forecast water table data (proactive) were used to determine wells requiring remediation, fewer wells would have been mitigated because 2008 to 2010 March water levels were higher than the base case levels. The decision tree developed for this project seemed to successfully identify 100% of the wells requiring mitigation action when appropriate input data was available (pump set depth, total well depth, and an accurate forecast of groundwater levels resulting from dam removal actions) as would be expected. A general comparison of spatial annual baseline water levels and subsequent model forecasts of post drawdown and dam removal seasonal low water table conditions did effectively identified 80% of the water supply wells requiring groundwater level response mitigation in the first year (Y1 stage 2 forecast). Subsequent models all identified 77% or more of the wells originally identified as possibly being impacted with up to 99% of the wells requiring mitigation identified by the Final Y3 modeling effort. This was not unexpected as the Y3 model was calibrated to well responses from all of the staged drawdowns. The goal of the project managers was to provide consistent supplies of water to every resident without noticeable gaps. In a few circumstances, wells became inoperable after they were recommended for mitigation due to lack of availability of well drillers (there were only 3 local drillers who were often times all working on other properties). The goal was met with temporary mitigation measures by piping water in from wells on adjacent properties to the households until the new well was completed. Water supply losses to individual residences were not reported to exceed 36 hours and in general never exceeded 6 hours. This was the true test of success for this project and was reflected in the positive public perception of the entire mitigation process.

6.3 Recommend Process for Mitigation Planning

As a dam removal is planned, a groundwater level impact analyses, and if needed, a mitigation process should be implemented. These actions require a number of steps:

1. Establish reference groundwater levels: A base case set of groundwater level conditions has to be established. For the Milltown study, historical and project gathered data on regional surface water groundwater interactions and flow systems were collected. This process required community support, planning, manpower, time, and hydrogeologic expertise. The extensive data sets provided an invaluable resource to use for numerical model creation and calibration. In retrospect, transient and continuous data both prior to and following the dam removal were necessary. Having pre dam removal groundwater level and river stage monitoring data provides a baseline to compare with post removal data which may avert potential litigation issues.

2. Forecast post dam out groundwater levels: Water table changes during and after dam removals need to be predicted. For the Milltown study a standard extensively calibrated numerical groundwater model was used. Our model was calibrated with historical, steady state and transient data sets. Models reproduced observed historical and initial monitored water levels with minimal residuals and the pre modeling groundwater budget suggests models were representative of hydrogeologic conditions. Forecasting future water levels is highly dependent on this calibration process and must also include observational data throughout a drawdown period to adequately evaluate system responses to engineering activities. Forecasts also need to be reported with some degree of uncertainty framed by using alternative conceptual modeling (model averaging) or geostatistical approaches. Reasonable forecasts of groundwater levels are a critical component of the decision tree input data and must be acquired through modeling or similar techniques. It has been proposed that simplified Q/K modeling (Berthelote, 2013Chapter 1) or Artificial Numerical Modeling (Berthelote, 2013Chapter 3) may be useful alternative tools for forecasting future groundwater levels, both of which require a more limited input data set.

3. Determine impacts: Simply predicting future groundwater levels resulting from a reservoir drawdown or dam out scenario does not allow decision makers to assess the risk of the water levels dropping below a well bottom or pump intake elevation. Well information (total depth or well intake and pump set depth) must be obtained in a data collection phase. For the Milltown study, well drillers logs and hundreds of well site visits were used to compile these data. Wells with the greatest potential threat (closest to the reservoir or shallowest well depths) were investigated first.
4. Develop a Mitigation Strategy: Threats of reduction or loss of productivity from individual wells were ranked. Once the individual well data were gathered and the predicted impact quantified (model results), then a recommended action was derived to assist managers in proactively mitigating wells with the greatest risk of failure from dam removal activities. A more descriptive explanation of this strategy is presented next.

This research resulted in the development of a risk based framework for proactive well mitigation necessitated by alterations to groundwater levels from a dam removal. We believe these approaches will be applicable to similar dam removal projects where well production losses are possible. The specific process used throughout this research (Figure 7B) can be reconfigured and modified to fit different sets of hydrological conditions (Figure 13). It is important to remember that while no approach will guarantee 100 percent protection all of the time, effective risk management reduces risks and increases the feasibility and effectiveness of remedial control or preventative options. Redundancies should be built into the system wherever feasible. These actions will mitigate repercussions when, and if, failures occur in the system and also help demonstrate that the mitigation managers have acted with due diligence.

![Risk management framework for a proactive dam removal well mitigation plan](image)

**Figure 13** Risk based decision process for groundwater mitigation resulting from dam removals.

Milltown managers used the decision tree presented in Table 2 to prioritize risk into manageable actions. In its most basic form, a risk assessment can simply be a ranking of hazards against designated benchmarks for the protection of consumers. Many standard risk matrices exist in different contexts. A mitigation plan reconfirms objectives (outlined in Figure 7) that were chosen for assessment as management targets or goals against which management actions will be evaluated. Making decisions that benefit stakeholders while maintaining watershed objectives can be challenging. The decision-making process is multidisciplinary in nature and must integrate variables such as scientific, socioeconomic, and political knowledge. All mitigation actions considered the risk framework, public perception, expert opinion, and cost benefit analyses.

In the Milltown mitigation process, regular public forums were conducted where forecasted impacts to groundwater levels and mitigation actions were presented to and discussed with the public. These public meetings generally started with discontent, suspicion, and misunderstanding but ended with cooperation, mutual understanding, and collaborations. Effective risk communication ensures all participants adequately understand the risk management process and how decisions are made. Educational activities executed during the Milltown dam removal included resource materials, seminars, workshops and public meetings.
Educational goals were to encourage awareness, understanding, and more informed decision-making. Public participation is the process by which all interest groups (stakeholders and the general public) in a community are provided the opportunity to make their views known.

7.0 SUMMARY

It is recognized that dam and reservoir removals will have some level of impact on adjacent aquifer systems. Managing groundwater system responses to dam and reservoir removals and the resulting economic and sociological consequences requires development of time effective mitigation strategies informed by groundwater level forecasting. In western Montana, project managers of the recent 8.5 m high Milltown Dam and the contaminated reservoir sediment removal implemented a well replacement mitigation strategy that attempted to proactively mitigate groundwater supply impacts from project activities. At the initiation of dam removal activities an extensive groundwater and river system monitoring network was established to observe groundwater level conditions and changes. A suite of multi-layer, three dimensional, finite difference groundwater models calibrated to both historical data and observed groundwater responses to staged reservoir drawdowns provided groundwater level forecasts of post dam out conditions. These data were inputs to a decision tree that identified wells needing replacement, lowering of pumps, further data collection, or were considered not at risk. Uncertainty was evaluated using sensitivity analyses and resulting alternative conceptual models. Observed results showed water table changes were up to 3 m near the reservoir and impacts extended 6 km downstream and about 2 km upstream of the reservoir. Model forecasts of groundwater level changes were greater than post dam removal observed levels, as expected, because natural variations in annual river driven groundwater recharge that occurred during the dam and sediment removal period were not used as model input. Risk tree analyses showed up to 115 wells were at risk. Proactive mitigation included the construction of 80 new wells and lowering of 20 pumps. Future dam removal projects should include groundwater impact analyses and proactive groundwater supply mitigation as needed. This process should include both observations of changes and pre-dam removal forecasting of groundwater responses. Forecasts should then be assessed and actions taken based on a logical project based decision tree such as the one developed here.

REFERENCES


Envirocon, 2006, Remedial design work plan, Final, For stages 1,2, and 3 drawdowns and related construction activities Milltown Reservoir Sediments Site.


Appendix 2A  Data For Figure 3 (Timeline) (Excel file available in digital format only)
Appendix 2B  Compiled Water Level And Climate Data For Study Period (Excel file available in digital format only)
Appendix 2C Final Calibrated MODFLOW Model Input Files (Files available in digital format only)
Appendix 2D  Well Survey Data (Excel file available in digital format only)
Chapter/Paper 3

Berthelote, Antony, Doctor of Philosophy, May 2013

Geosciences

The Role of Drawdown Data in ANN Forecasting of Water Table Responses to Dam and Reservoir Removals

Chairperson: Dr. William W. Woessner

ABSTRACT:

Planning for and mitigating a groundwater system response caused by dam and reservoir removals requires the development of methods to forecast post dam removal groundwater levels. A standard approach to generate the required data involves the development of extensive field based hydrogeological data sets and the application of sophisticated mechanistic models that solve for the three-dimensional distribution of fluxes and heads. Such models require large amounts of costly data and typically require long run times. An alternative is to use statistical models that capture the relationship between surface processes and the response of the groundwater system. This research assesses if Artificial Neural Network methods (ANN) can be used to forecast groundwater level changes likely to occur from a dam removal action. A groundwater level response data set obtained during the removal of the Milltown Reservoir in western Montana was used to assess ANN model performance. To further evaluate the ANN modeling forecasts, results were also compared with forecasts made with a three dimensional MODFLOW deterministic model. ANN modeling was conducted using MATLAB software and associated tool boxes. ANN forecasts of groundwater levels utilized daily river discharge, temperature, and sets of field measures of pre reservoir drawdown reservoir stage (ANN model AM1). However, ANN forecasts were improved by including groundwater level data collected during a partial reservoir drawdown (ANN model AM2). The ANN model trained without the reservoir pool drawdown signal (AM1) residuals for the two subsequent staged reservoir drawdown forecasts were 1.1 m and 5.8 m. Model AM2 produced a residual of 0.7m for both forecasts. Average Root Mean Squared Error (RMSE) for AM1 and AM2 forecasts over a two year forecast period were 2.1 m and 0.7 m. In comparison, the RMSE for the calibrated deterministic model were 1.3 and 1.4. It was concluded that AM2 ANN modeling produced post dam out groundwater level forecasts that were similar to both the deterministic model results and field observations. It is suggested that ANN groundwater level forecasting inclusive of training data containing a preliminary or temporary reservoir pool drawdown will provide managers with a reasonable representation of post dam out groundwater conditions.

KEY WORDS  Groundwater Level Forecasting; Artificial Neural Networks (Anns); 3D Numerical MODFLOW Modeling; Milltown Reservoir; Dam Removal Mitigation And Management; Groundwater Surface Water Interactions

1.0 INTRODUCTION

There is increasing interest in removing dams in the United States to remedy adverse ecological impacts, eliminate risks associated with the deteriorating conditions of aging dams, and address societal pressures to restore rivers to more natural settings (Babbitt 2002; Farinacci 2009; Hart et al. 2002; Landers 2004; O’Conner, Major, and Grant 2008). Dams and associated reservoirs provide a variety of economic, environmental, and societal benefits, including recreation, flood control, water supplies, hydroelectric power, waste management, river navigation, and wildlife habitat. In the United States, there are approximately 2.5 million small dams less than 1.8 m high, 80,000 large dams over 1.8 m high, and 8000 major dams greater than 15.2 m high (American Rivers, Friends of the Earth, and Trout Unlimited 1999; Bowman et al. 2002; USACE 2008). To date, dam removal studies have focused principally on geomorphologic changes (Doyle et al. 2003; Evans et al. 2000; Graf 2003; Hart et al. 2002; Collier, Webb, and Schmidt 1996). Dam removal projects are commonly planned without consideration of the likely changes in groundwater conditions that will occur in adjacent aquifers.
Industry, agriculture, and population centers are often established in close proximity to dam sites when a reservoir is created. One in four of the major dams in the United States are constructed in settings with underlying and/or adjacent unconsolidated and semi-consolidated sand and gravel aquifers. Groundwater from these systems is often used for primary water supplies (U.S. Dept. of the Interior, 2008; USACE, 2008). A few investigators have reported that after reservoir creation groundwater levels in the adjacent landscape would locally increase (e.g. Leopold and Maddock, 1954). Heilwell (2005) documented the presence of additional groundwater in a fractured bedrock system after the construction of a reservoir. It can therefore logically be presumed that based on hydrogeological principals and the literature that, in most settings, groundwater levels will typically rise underneath and adjacent to newly constructed reservoirs. Conversely, it is reasonable to expect that groundwater levels will decline to their previous levels following a dam removal. In response, water level declines have the potential to impact existing groundwater use and groundwater supported ecological systems that have developed over the life of the reservoir.

Interestingly, the body of published literature addressing both observed and predicted groundwater changes during and after dam removals is extremely limited. Recent pre-removal environmental assessments have begun to mention possible well failures from groundwater head loss following reservoir pool removals (e.g. the 2.75 m high Finesville Dam (Musconetcong River in NJ) (USDA, 2010) and 11.2 m high Gold Ray Dam (Rogue River in OR) (NMFS, 2010)). However, “after the fact” well mitigation is more the norm. The 39 m high Condit Dam (White Salmon River in WA) Environmental Impact Statement predicted “... no significant unavoidable adverse impacts...” to groundwater resources. However, in October 2011 well mitigation was being discussed to replace multiple failed wells following the dam removal (Sandison, 2010). Wyrick (2009) outlined the social concern for well mitigation from the 6 m high Wadsworth and 4.4 m high Sterling Lake in the Mantua Creek Watershed.

The lack of literature on groundwater impacts resulting from dam and reservoir removals may, in part, be because few large dams have been removed up to this time. With the exception of water table changes resulting from the removal of small scale beaver dams which are only a meter or two in height and of limited areal extent, no comprehensive pre- and post-dam groundwater response studies have been reported in the literature (Butler and Malanson 2005; Chen and Chen 2003; Mertes 1997; Naiman, Johnston, and Kelley 1988; Westbrook, Cooper, and Baker 2006; Woo and Waddington 1990). To avoid future conflicts with water users adjacent to reservoirs planned for removal, resource managers will need appropriate methods to forecast the consequences of dam removals on groundwater systems.

Standard groundwater modeling approaches include analytical and numerical methods. An ideal model would produce a reasonable prediction (based on the post removal groundwater level data) at a degree of uncertainty that is appropriate for the related management action (e.g. identifying the number of domestic well replacements or remediation, and/or the economics of increased pumping costs). Analytical models are typically designed to represent relatively simple space and time causes and effects; thus, in most dam removal settings, they would be of limited value (Guo, 1997). Deterministic numerical models such as MODFLOW (Anderson and Woessner, 1992; Harbaugh, 2005) and FEFLOW (Diersch and Kolditz, 1998) allow for the representation of complex hydrogeologic settings and conditions, and can simulate steady state and transient condition in two or three dimensions. Such models require extensive field derived physical and hydrogeological input data. When extensive data sets are available and model calibration can be completed, deterministic modeling approaches are likely to provide adequate tools for post-dam removal groundwater level forecasting (Berthelote, 2013). Alternatives to analytical and deterministic modeling approaches include geostatistical (Wiese and Nutzmann 2011; Diodato and Ceccarelli 2006) and non-linear time-series analysis methods. Our interest here is to assess if Artificial Neural Networks (ANN) modeling forecasts of groundwater responses to a dam removal provide a reasonable alternative approach to standard numerical deterministic methods. Citations of the success of ANN modeling in other disciplines report their use as a more practical and cost effective alternative to predicting outcomes than complex deterministic modeling approaches. Applications of ANN modeling of groundwater levels driven by environmental and climatic stresses have recently been examined in a large scale multi-level confined aquifer system, a lake recharged aquifer system, to represent dynamic head boundaries in an arid environment, and the impacts of pumping and possible changes in climatic conditions on water table elevations in a semiconfined glacial aquifer (Coppola et al., 2005; Dogan et al., 2008; Huo et al., 2011; Nourani et al., 2008).
Given that ANN has been used successfully to forecast groundwater levels in a variety of steady state systems, we investigate its potential to forecast groundwater levels following a large system change, the response of a groundwater system to the removal of a dam and reservoir. Unlike the previous research using ANN forecasts, once a reservoir is removed, the entire system is altered resulting in potentially different surface-subsurface responses. The focus of this research is in three areas: first, assess the capability of ANN modeling to adequately forecast groundwater responses to large system changes (dam removal) as an alternative to applying industry three dimensional deterministic numerical modeling approaches; secondly, assess the value of training data (reservoir pool drawdown signal) that approximate the system change; and thirdly, propose ANN modeling strategies for forecasting responses of groundwater systems to dam removal actions. Hydrological data sets and deterministic modeling of the groundwater response to the removal of the Milltown Dam and reservoir complex in western Montana provided the observational data and comparative forecasts by which ANN modeling was evaluated. Specifically we hypothesized that an ANN model trained with pre dam out groundwater level data alone will poorly predict post dam out groundwater levels. In addition, we hypothesized that groundwater level data used for training the ANN model that also captures a partial or temporary pre dam out reservoir drawdown data set will provide predictions that more closely match observations and deterministic model forecasts.

2.0 STUDY SITE AND BACKGROUND
The groundwater response data were generated during the three year Milltown Dam removal project located 8 km east of the city of Missoula, Montana (Figure 1). The study area extends from Hellgate Canyon at the eastern edge of the Missoula City limits upstream 13 km to Turah Bridge in western Montana (Figure 1). The Milltown Dam was located at the confluence of the Blackfoot and Clark Fork rivers at the center of the study area. The reservoir extended approximately 1.5 km upstream and had a full pool width of approximately 0.75 km (Berthelote et al., 2007). The Milltown area has a semi-arid climate with a mean annual temperature is 13.7 °C and the mean annual precipitation of 35.1 cm (http://www.ncdc.noaa.gov). The average discharge of the Clark Fork River 4.5 km downstream of the reservoir (USGS gauge number 12340500) is 83.2 m³/s (USGS, 2011).
The Milltown Dam construction was completed in 1907 at the confluence of the Clark Fork and Blackfoot rivers. Over the next 100 years, the Milltown reservoir filled with mining and smelter wastes from the Butte and Anaconda area located 140 km upstream. The Milltown Reservoir was designated a CERCLA (U.S. EPA Superfund) site in 1983 as water seeping from the reservoir sediments recharged the adjacent coarse-grained aquifer and contaminated local wells with dissolved arsenic, iron and manganese (ARCO 1992; Harding Lawson Associates (HLA) 1987; Moore and Woessner 2002; Udaloy 1988; Woessner et al. 1984).

Residents located adjacent to and proximal to the reservoir utilize the bedrock bounded 6 to 60 m thick coarse sand, gravel, and boulder unconfined valley aquifer for all domestic and community water supplies. Aquifer recharge is principally from four sources: 1) river channel leakage (losing channels of the Blackfoot River and the Clark Fork River (below the dam to Hellgate Canyon); 2) leakage from the Milltown Reservoir pool; 3) lateral valley underflow at the Turah Bridge and Blackfoot Canyon boundaries; 4) limited inflows at the bedrock boundaries. Groundwater discharges principally as underflow through the valley aquifer located in Hellgate Canyon to the west, and locally in gaining stream sections of the Clark Fork River (above the reservoir and, at one time, just below the dam). The groundwater flows towards the reservoir area from the upper Clark Fork River valley and converges just above the reservoir with the groundwater entering at the mouth of the Blackfoot River canyon, then flows northwest down valley (Figure 1). Aquifer flow rates are measured in 10's of m/d, the coarse grained nature of the sediments allows rapid flow. Hydraulic conductivities range from 90 to >27,000 m/d and reflect the high energy deposition environment and coarse grained nature of the sediments. With measured horizontal hydraulic gradients 0.0013 (upper Blackfoot River arm) to 0.066 (near well 05) groundwater velocities are measured in 10’s of meters per day (porosity estimated at 0.20). Vertical riverbed hydraulic conductivities range from 0.43 to 12.8 m/d and groundwater river channel exchange rates range 0 to 4 m$^3$/(day $m^2$)

Figure 1: Study area location map. Blue arrows represent groundwater flow directions. Well identifiers are the sites corresponding with the well responses forecast by ANN modeling. Well 03 data was incomplete and therefore removed from subsequent analyses and comparisons.
In 2004 the U.S. EPA, the State of Montana, and stakeholders decided to remove the Milltown Dam and 1.9 of the 5 mcm (million cubic meters) of contaminated reservoir sediments. The 8.5 m high Milltown Dam and the associated reservoir were removed during the period of 2006 to 2009. Remediation and restoration plans were designed to be completed in stages (Environco, 2006a, b; Westwater Consultants et al., 2005). This estimated $100+ million remediation/restoration project required three drawdowns starting in 2006 (3.5 m in March 2007, 3.5 m in March 2008, and 1.2 m in April 2009) that correlated with engineering tasks prior to reaching the final free flowing state in 2009. Before the dam removal process formally began, project engineers initiated a 3.5 m temporary drawdown in November of 2005 to examine the submerged portion of the dam. It was observed that groundwater levels in some wells adjacent to the reservoir declined and a few shallow wells became inoperable. These conditions resulted in the initiation of an expanded water level monitoring network, and the construction and calibration of an industry standard three dimensional deterministic numerical groundwater model (Berthelote et al. 2007; Berthelote, Woessner, and Thompson 2010). A MODFLOW model was setup in 2006 and transiently calibrated throughout the initial drawdown. This model was used to forecast likely groundwater responses to reservoir stage drawdowns and complete dam and reservoir removal. The MODFLOW model forecasts combined with a decision tree resulted in remediation actions (Berthelote, 2013). The State of Montana and U.S. EPA implemented a well replacement and well mitigation program that attempted to limit water supply impacts before further reservoir stage declines were implemented. The numerical groundwater model was updated annually with new observation data and used to re-evaluated potential groundwater level changes resulting from planned drawdowns. It is the field observational data and numerical modeling results from this previous effort that will be compared to the ANN modeling output described in this work.

### 3.0 METHODS

We evaluated the performance of the ANN by examining to what degree it could forecast the observed groundwater levels prior to and after the dam removal. We used variations in the composition of the datasets used to train the ANN. We were specifically interested in evaluating if the inclusion of preliminary reservoir drawdown information in the ANN training process improved the forecasts of post dam out groundwater level predictions. The ANN predictions were also benchmarked against the results of the deterministic modeling performed with MODFLOW (considered an industry standard mechanistic groundwater model).

#### 3.1 Observed Water Level Changes

Groundwater levels were monitored using recording transducers and electric hand operated water level monitoring tapes (Berthelote et al. 2007; Farinacci 2009; Tallman 2005). Data derived from a 74 well network was combined with historical non-continuous hydrologic data dating back to 1981 and compiled into a single database containing 226 individual wells with over 2000 measurement days (Berthelote et al., 2010). Farinacci (2009) recognized that approximately 80% of the observed response of the water table was controlled by the river stage/discharge conditions. Therefore, the observation database included the reservoir pond and tailrace elevations (North West Energy, 2007) as well as the available USGS river stage data (USGS, 2011). The timing and magnitude of the unconfined groundwater system response to reservoir stage and dam removal activities was determined by analyzing pre- and post-dam removal groundwater level trends at wells located throughout the study site (measurement error of 0.02 m) (Berthelote et al., 2007).

#### 3.2 Three Dimensional MODFLOW Model

More than two decades of hydrologic investigation of the Milltown Reservoir Superfund site resulted in the construction of two earlier two dimensional numerical groundwater models; however, only portions of the study area were modeled (Brick, 2003; Gestring, 1994). Managers and regulatory agencies responded to well failures induced by the 2005 temporary drawdown by funding an integrated three year extensive field data collection campaign and development of a three dimensional groundwater model to forecast the magnitude, timing, and location of likely physical changes in the groundwater system (Berthelote et al. 2007; Berthelote and Woessner 2008, 2009; Berthelote, Woessner, and Thompson 2010). This model was used to inform mitigation planning. The model was updated and adapted as drawdown and dam removal...
activities progressed, operation scheduling changed, and new observational data became available. As a consequence of the dynamic nature of this large de-construction effort, forecasts were adjusted and revised as new information became available. Complete documentation of the model development, calibration, forecasts and uncertainty analyses are presented in the referenced documents (Berthelote et al. 2007; Berthelote and Woessner 2008, 2009; Berthelote, Woessner, and Thompson 2010). A summary table of groundwater and surface water model inputs can be located in Berthelote (2010) Appendix B. The final six layer three dimensional numerical MODFLOW model (Berthelote et al., 2010) consisted of 53,192 active 46 m by 46 m cells. Using standard techniques, it was parameterized and calibrated to steady state conditions (March 31, 2006), transient conditions (March 31, 2006 to May,9, 2010), and history matched with October 8, 1992, steady state data and a second transient data set, 1992-1993 (e.g. Anderson and Woessner). Calibration used automated Least Mean Squares analyses and root mean square error (RMSE) analysis to evaluate model fit. It should be noted that calibration of this mechanistic model to observational data required data inclusive of system responses from initial and subsequent drawdowns. Well level responses, river fluxes, changing reservoir configurations, and pool level changes were among the data sets needed to complete calibration of the changing system.

3.3 Application of ANN to the Milltown Study
We used MatLab software and the associated neural network toolbox (http://www.mathworks.com) for our ANN modeling (Figure 2). We trained two networks with identical architecture but with different training datasets. Each model was executed using data representative of a period where both inputs and the groundwater response (measured change in water level at a monitoring well) were known

A two layer feed-forward network with Levenberg-Marquardt back propagation learning algorithm (trainlm) was utilized. Two non-linear transfer functions (Tan-Sigmoid Transfer function and Purelin Transfer function) were applied to layers one and two respectively. Our approach was to develop the model using the most parsimonious (lowest number of neurons) network architecture needed to achieve satisfactory results. This was done to decrease the chances of data over-fitting. We started with an ANN structure with one node in the hidden layer and increased the number of nodes by 1 in each successive runs until the ANN was considered successfully trained or the number of nodes required for a successful run exceeded sixteen. We considered the network trained if the RMSE of the predictions was better than 0.75 m in less than 200 iterations. Acceptable convergence was generally achieved in fewer than 100 iterations. Performance goals were determined based on behavioral observations for each well and averaged 0.23 m for all 12 wells. A copy of the ANN Code used for this research is located in Appendix 3A.

3.3.1 Data Set
For the Milltown site, observed spatially distributed variations in groundwater levels at 12 wells were used to assess ANN predictions of drawdown impacts for each of two ANN model architectures. A total of 731 days of input training data were used. Input training data sets included two years of daily measurements beginning a month prior to the initial reservoir drawdown (March 2, 2006) to just before the second drawdown (March 01, 2008). This enabled the training/validation model to capture groundwater level behavior impacted from drawdown activities, required for the testing/forecast model to mimic the impact during each staged drawdown and future dam out river conditions. Training and validation used 80% and 20% of the available input data, respectively, interlaced for the time period examined.

The observed groundwater level data described above were initially subsidized with reservoir pool level data (North West Energy, 2007) and potential ANN modeling inputs derived from nineteen continuous
climatologic- and hydrologic-data sets acquired from two public internet sources (http://wrcc.dri.edu/wraws/ and USGS). Climate data was acquired from the Missoula FTS Montana RAWS data station located at Latitude 46° 51’ 00”, Longitude 114° 03’ 00”, 15.4 kilometers from the site (at an elevation of 976 m). Hydrologic data were acquired from the USGS real time stream data database for station number 12345000. Independent pre-processing used linear cross correlation to remove input parameters that had minimal (<0.75 correlation coefficient) or no signal contribution to the groundwater levels observed for the training period. The final three selected inputs were reservoir pool level, river discharge, and air temperature. These input data were normalized to a mean of 0 and a standard deviation of 1 prior to the ANN analysis to avoid undue influence of data types with relatively large values (magnitude) and variability during the training process.

3.3.2 **ANN Input Selection Method**

Two ANN models were constructed to evaluate how the nature of the training data, specifically how the absence or inclusion of groundwater responses to a reservoir pool drawdown event, impacted forecasts of groundwater level responses. The two models used identical inputs for training, validation, and forecasting with one exception. AM1 training used reservoir pool level data that did not include a reservoir drawdown signal AM2 training alternatively used pool level data inclusive of a drawdown signal during the training and validation period. For both models a forecast was conducted using the same 3 input types used for training and validation where a pool level step function representative of the actual total staged reservoir drawdown for the forecast period was substituted for the pool level inputs for both models. The other two inputs were simply duplicated for the forecast period (assuming that the river discharge and temperatures represented a steady state system and that no future data would be known). The two forecast responses of groundwater levels to the dam out conditions allowed for a controlled experiment evaluation of how the use of drawdown data during the training and validation process impacted predicted groundwater responses.

Table 1 illustrates the variations in the data sets used for each model:

| Table 1 | We trained two networks with identical architecture but with training datasets varied by one input. This table identifies the input data used for the two models AM1 and AM2 (Appendix 3B). Inputs were river discharge (Q), daily temperature (T), and reservoir pool level represented by a full pool (P₁) or reservoir pool levels inclusive of a staged drawdown (P₂). Head or water table elevation in each well is represented as h. AM1 and AM2 varied only by input 3 during training and validation but both used a step function to represent the staged drawdown of the reservoir pool (P) for model forecasting along with the original Q and T data used for training.

For this modeling evaluation, since the ANN models were also being evaluated to determine how well they forecast groundwater level responses to reservoir drawdown and dam removal actions at Milltown, the forecasts were compared with observed groundwater responses. In addition, ANN modeling results were also compared and contrasted with three dimensional groundwater modeling forecasts to assess prediction similarities or differences.

3.4 **Statistical Methods**

Three techniques were used to evaluate performance of the two ANN models relative to observed and deterministic model groundwater levels: 1) a subjective visual hydrograph comparison of residuals of ANN
model transient forecasts and groundwater level observations; 2) a standard root mean square error (RMSE) analyses of the residuals used to evaluate fit for transient paired data:

$$\text{RMSE} = \sqrt{\frac{1}{P} \sum_{i=1}^{P} [(x_m)_i - (x_s)_i]^2}$$

where subscripts $m$ and $s$ represent the measured and simulated outputs, respectively, and $P$ is total number of events considered; and 3) a T-Test evaluation of the model residuals to evaluate the statistical relevance of the reservoir pool level input.

## 4.0 RESULTS

### 4.1 Observed Data

Natural historical groundwater level fluctuations proximal to the dam seasonally vary 1 to 2 m. Below the dam site, water table elevations typically vary annually up to 4 m, whereas, above the reservoir site annual variations in groundwater levels are usually less than a meter. Well hydrographs mimic stream stage changes with the highest groundwater levels occurring during snowmelt driven high river stages in late spring and low water levels corresponding with late winter low flow conditions. Farinacci (2009) observed groundwater responses to changes in river stage occurred with little or no time lag.

Figure 3 Spatial representations of the observed groundwater changes (background shaded map) from March 31 2006 to March 31 2010. Graphs illustrate the observed data (blue lines) in Wells 12, 05, and 10 below, at, and above the Milltown Dam, respectively. Red boxes and connected dashed highlight the occurrence of low groundwater levels in March each year and the annual variation in this level from year to year.

Groundwater levels were also impacted by reservoir drawdowns during the reservoir drawdown and removal process (Stage 1 drawdown 3.5 m; Stage 2 3.5 m; and Stage 3 1.2 m) (Figure 3). The impact of the first drawdown in June 2006 coincided with the natural declining limb of the hydrograph making reservoir drawdown only impacts to groundwater less obvious (possibly an increased slope in the hydrographs). The remaining drawdowns were scheduled to coincide with the historical March low river flows. Once again separating reservoir drawdown groundwater responses from well hydrographs is partially masked by antecedent groundwater recharge conditions that also influenced March groundwater levels. When the entire groundwater level records are reviewed and the changes in March 31 water levels used as reference points, reservoir removal appears to account for about a 3 m reduction in groundwater levels proximal to the dam.
4.2 MODFLOW Modeling Results

The most parsimonious model became the 244 cell by 169 cell 6 layer MODFLOW model with 53,192 active cells. The model required input parameters (conductivity, recharge, initial head, etc.) for every stress period and layer at each cell location. Additional parameters for each boundary condition cell were required (head boundaries, river cells, drains, wells, etc.) resulting in more than 783,000 input parameters. Detailed information on model construction, calibration and uncertainty analyses are reported in the work of Berthelote et al. (2007, 2008, 2009, 2010). The groundwater levels (modeled verses observed residuals were generally less than 2 m for each of the network wells used in calibration) generated by deterministic modeling were considered to be an adequate representation of groundwater conditions both before dam removal and after dam removal. Model calibration suggest that MODFLOW results appear to generally under predict the highest and lowest portions of the groundwater level trends at some locations.

Predicting future conditions has many challenges as both hydrogeological and stream elevations, and river bed leakage properties and river stages (river discharges and durations) are unknown but required forecast input parameters and must be approximated. In addition, the construction of the calibrated numerical model is not a complete representation of the complex hydrological system which is an unfortunate artifact of all mechanistic models that try to mimic complex systems. However, the aggregate of computed post dam removal forecast ranges provided managers with sufficient information to assess groundwater level impacts and develop a proactive remediation program (Berthelote and Woessner 2009; Berthelote, Woessner, and Thompson 2010). Each model forecast provided groundwater levels for single day in the future that represented the groundwater response to a set of new conditions (reservoir level drawdowns). As stated previously, new groundwater level data acquired during the three year dam removal phase was continually used to update, refine and recalibrate the working model. Both the simulated heads of the final deterministic calibrated model and the two forecast head distributions from the initial deterministic modeling were used for comparison to the ANN results presented below.

4.3 ANN Modeling Results

Compared to daily observed or monthly deterministically simulated results, ANN solutions tended to produce higher frequency signals around a central trend line. However, ANN results captured observed temporal and spatial spring groundwater level peaks and winter declines (Figure 4). Mechanistic or other spatial modeling suffer from compounding errors due to parameter uncertainties, interpolation, extrapolation, or other similar techniques. Since ANN models do not utilize this approach they are not affected by these types of compounding errors.

The well hydrograph shown in Figure 4 illustrate comparisons of the observed and forecasted results for well 01 that is closest to the Milltown Reservoir. This well location captured the largest magnitude of water level changes.

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Steady state AM2 results for stage 3 (Dam removed) forecast maintain a mean residuals of 0.4 m, which is comparable to the 0.7 m residual of the initial MODFLOW model forecast and significantly better than the AM1 residuals of 2.2 m. Transient results for the period 2008 to 2010 maintained similar performances with RMSE values for AM1, AM2, and MODFLOW of 2.1 m, 0.7 m, and 0.5 m. The Transient MODFLOW values however do not reflect a true forecast and are representative of the final calibrated model results with all data for the period included in the model. Similar residual comparisons are
documented for the 12 individual study wells in Table 2 demonstrating the effectiveness of ANN modeling in capturing general groundwater level trends during the Milltown staged reservoir drawdowns and subsequent complete dam removal. The maximum RMSE’s for the 12 wells tested were 5.2 m and 1.1 m for AM1 and AM2, respectively and maximum residuals for steady state forecasts were 5.8 m and 1.2 m respectively. Individual well RMSE values for AM2 that met or exceeded 1m (wells 01, 05, 12, & 13) were the closest wells to the reservoir area or down river from the dam. All the other wells were less impacted by drawdown events as they were upstream of the main reservoir or closer to the valley boundaries less proximal to the river. A plot of the distance from surface water (reservoir pool or river) vs RMSE or variance (from the t-test) demonstrated that ANN modeling residuals for individual wells increased proportionally with the proximity to surface water (Figure 5). This is a function of signal dampening produced by the subsurface.

Statistical residual comparisons of RMSE, standard deviation, and mean values for ANN models all demonstrate that AM2 which included reservoir and groundwater drawdown training information produces forecast groundwater levels that are more representative of observations than did the AM1. The t-test results comparing the absolute values of the residuals for these two models with a significance level set to 0.001 demonstrates this fact with an average calculated p value of 3E-23 with individual well p values ranging from 8E-81 to 4E-22. The fact that the p value is far less than the significance level and is approaching zero concludes that the observed effects were unlikely to be the result of chance alone and that the null hypothesis is false (Goodman, 2008). The AM2 results are therefore statistically significant.

5.0 DISCUSSION
The literature suggested that ANN modeling could be used to forecast hydrologic time series (Dogan, Demirpence, and Cobaner 2008; ASCE and The Task Committee on Application of Artificial Neural Networks in Hydrology 2000), including groundwater level forecasting (Dogan, Demirpence, and Cobaner 2008; Nourani, Mogaddam, and Nadiri 2008). However, its application to forecast the response of groundwater levels to dam and reservoir removals is a new application. Resource managers and regulatory agencies overseeing dam removal projects need to consider how actions will affect adjacent groundwater systems. This responsibility can be addressed using two basic methods, monitoring or modeling. It can be argued that establishing both spatially and temporally pre-dam and reservoir removal baseline hydrogeologic data (groundwater levels and river and pool stages) is a critical step in assessing the impacts of removal actions on the adjacent groundwater resources. Observing groundwater levels and stage changes during and after dam removal processes will likely result in mitigation following the removal process (reactive approach). A more proactive approach would be to integrate pre dam removal hydrogeologic information and forecast water level changes resulting from dam removal plans. This approach allows for the development of mitigation plans and allows execution of plans prior to observing impacts.

Forecasting likely groundwater level changes requires some degree of groundwater modeling. Physically-based deterministic numerical groundwater models such as MODFLOW are widely used to identify impacts of natural and human-induced changes in the subsurface environment. Such models enable us to conduct a series of numerical experiments to analyze subsurface flow and transport phenomenon under
varying physical, biological and chemical processes. However, because of the practical difficulties of representing all the natural subsurface complexity, model results include a degree of uncertainty. The implications of these uncertainties are particularly significant when the models are used in practical applications for prediction or extrapolation purposes under varying environmental conditions (Demissie 2008; Mohammadi, Eslami, and Qaderi 2008). This modeling methodology includes costly data requirements which can take years of data discovery and interpretation. Finally, forecasts require a representation of future conditions including groundwater recharge rates and timing, river stage changes, and groundwater discharge rates and timing, and changes in stresses such as pumping. Accuracy of forecasted groundwater level responses is dependent on how well future modeling scenarios match reality. Additionally, if the entire system is altered as in the case of a dam removal, model forecasts can be poor if they do not integrate some observational changes of initial system responses to changing conditions (observations measured during a drawdown event including; changes in fluxes from altered river or reservoir configurations, groundwater level responses etc).

Our data support the hypothesis that using ANN to forecast groundwater levels following a dam removal in the Milltown setting is feasible when the training datasets include pool levels during a drawdown event. ANN results were compared to temporal and spatial observational data and contrasted with MODFLOW results. We concluded that ANN solutions are similar to those obtained using a standard numerical groundwater model in this case.

It was determined that transient data must contain an adequate representation of hydrogeologic conditions such as an annual cycle of water level change and, as stated previously, information on how the groundwater system responds to a change in reservoir stage (a response signal). In our case, this response signal data set was the observed groundwater level changes in response to the first stage of the reservoir drawdown. Such a data set is also needed to build an appropriate mechanistic model including a dam removal event. Ideally, to maximize the information from observing a drawdown the timing of a response signal data set should coincide with a hydrological period in which few additional factors are influencing groundwater level change. Unfortunately, at the Milltown site, the observed groundwater level data set representing the response to the reservoir level decline coincided with noisy background signals making training and calibration not ideal. Analyses of data correlations minimize the number of inputs for the ANN modeling. It also may be useful to precondition data inputs to minimize high frequency signals (low pass filter), a process that would result in smoothing ANN modeling solutions. Such data processing would likely avoid anomalous responses to high frequency signal inputs (e.g. large precipitation events and/or ice dam induced recharge events). It is also recognized that further data reduction techniques, such as dimensional analysis or principal component analysis may allow for maximization of information and minimize any redundancies like interaction effects between inputs.

It is not surprising that groundwater level forecasts are predominantly dependent on river discharge, climatic data that serves as proxies for snow melt and groundwater recharge timing (maximum and average temperature, solar radiation, and total heating degree days), and reservoir pool levels. ANN input parameters will likely change when attempting to predict groundwater level responses in other hydrogeological settings dominated by alternative sources of recharge and discharge. For example, a precipitation dominated groundwater recharge system would presumably utilize any number of precipitation inputs and be less dependent on temperature data. The advantage of the ANN methodologies presented here is that input selection can be semi-automated, allowing for the evaluation of a wide variety of data sets.

It should be cautioned that if one or more available inputs are independent or weakly related to the dependent variable, the ANN may have difficulty reaching a viable solution. The explanatory value of input variables is highly dependent on the type of system under consideration. For example, if a dam is removed in a low permeable (bedrock) or confined system where the aquifer recharge does not originate from the reservoir leakage, then: 1) the impacts to the local groundwater system would presumably be negligible or absent, and therefore 2) any attempt to use the ANN methodologies presented in this work would lack the correlations required to reach an acceptable solution. More importantly, due to the poor extrapolation power of ANN, any assessment of impacts under conditions beyond these represented in the calibration data set may be potentially subject to large errors. Standard numerical groundwater models (MODFLOW)
are better suited to predict complex systems where physical impacts in the system or where strong transient alterations of the inputs beyond calibration (training) conditions are expected. ANN solutions are limited to point location, and independent interpolation techniques are needed to reconstruct the spatial distribution of the water table. In contrast, depending on the space and time discretization, deterministic modeling provides solutions at every cell in its domain although it is important to keep in mind that these models are vulnerable to errors in the generation of the spatially distributed parameter fields needed to run them. Both modeling approaches require information about future conditions to allow for accurate forecasting. ANN modeling would need future climatic, river and pool stage data sets. Realizing that there is a large degree of speculation and uncertainty in these data, a reasonable approach would be to run a number of likely scenarios and create a forecast tempered with probability information. In our case, based solely on observed data it is not clear how the future March low water table positions will be modified by natural variations in the magnitude, duration and distribution of steam stage and discharge, and changes in the river bed sediment character (leakage properties). For example, it is likely that a series of drought years may limit stream/aquifer recharge and result in lower water table positions than measured in 2010, one year after the dam was removed.

Despite the limitations of ANNs, they are a convenient method that permits real-time continuous improvement of the forecasts as new data become available for further training and permits predictions and offer useful information even for poorly understood systems (Hertz 1991; Zurada 2006.; Sung 1998). This may provide decision makers with a convenient tool for managing water resources without the need for extensive data collection. Second, the increasing number of dam removals, particularly larger dams, will necessitate a rapid and cost-effective consideration of the impacts on local groundwater systems that ANN’s can provide.

6.0 CONCLUSIONS
Two potential ANN scenarios were developed and used to evaluate the importance of training data that contained groundwater level responses to reservoir drawdowns in providing post dam removal estimates of groundwater levels. AM1 forecast impacts to groundwater without a drawdown response training data set and AM2 utilized such information. AM2 forecasts more closely matched forecast generated from a MODFLOW model and observed groundwater levels than the AM1 models. The maximum RMSE’s for the 12 wells tested were: AM1 (5.2 m) and AM2 (1.1 m), with respective averages of (2.1 m and 0.7 m). The MODFLOW model required 3 years of extensive field data collection and iterative expert model calibrations to produce forecasts. In contrast the semi automated ANN solutions used data from 2 internet data sources (USGS and RAWS), reservoir pool levels acquired from the dam operation records, and at least one set of groundwater level data proximal to the dam for calibration matching. Though many environmental data sets are available for inputs into ANN models, they should be limited to those that have some correlation with aquifer recharge. Input conditioning must also include data normalization. We found that it would be practical to implement ANN modeling to forecast the response of individual and groups of wells to dam and reservoir removal actions. This approach would be useful to water managers and project leaders as the consequences of remediation and restoration are considered. The reduced data and manpower requirements for ANN modeling make it a practical methodology in hydrogeologic settings where a reservoir contributes to aquifer recharge for forecasting how groundwater levels are likely to change when a dam is removed.
REFERENCES:


Envirocon, 2006a, Milltown Reservoir Sediments Operable Unit Remedial Action Monitoring Plan, p. 263.

Envirocon, 2006b, Remedial design work plan, Final, For stages 1,2, and 3 drawdowns and related construction activities Milltown Reservoir Sediments Site.


| Appendix 3A | ANN Code (Files available in digital format only) |
Appendix 3B  ANN Input Files ANN Data (Inputs and Results) (Files available in digital format only)