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October 2008

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Prepared by:
Desert Research Institute, Reno and Las Vegas, NV
Colorado State University, Fort Collins, CO

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EXECUTIVE SUMMARY
Introduction and Objectives

There is little doubt that seepage from unlined water delivery canals occurs on a local and regional scale and that losses may be significant in particular areas. Traditional seepage-abatement technologies such as compacted earth, reinforced or unreinforced concrete, and buried geomembranes are typically used in situations where seepage rates are elevated and projected water savings offset their high construction and maintenance costs. However, as water resources become further constrained in the arid western United States, there is a need for cost-effective seepage reduction technologies that can be used in locations where traditional methods are cost-prohibitive. A cost-effective technology could also be used on a regional scale to address regional problems such as seepage-induced high groundwater tables or the mobilization of subsurface salts to surface water. The granular form of linear anionic polyacrylamide (LA-PAM) has been identified as one such technology capable of cost-effectively reducing seepage rates from unlined water delivery canals. Granular LA-PAM is one type of a broader family of polyacrylamides that has a variety of uses, including as a flocculant in wastewater treatment, in food packaging, and paper manufacturing. Over the last decade, polyacrylamides have found increased use as an agent to reduce erosion and sediment transport from crop fields and construction sites.

In 2005, the Desert Research Institute (DRI), in collaboration with the U.S. Bureau of Reclamation, initiated a series of field and laboratory studies to assess the benefits and risks of LA-PAM used as a method to reduce seepage from unlined water delivery canals. The primary objectives of this research were to:

1. Quantify potential seepage reduction benefits, water savings, and typical application cost.
2. Address potential risks of LA-PAM use emphasizing the downstream transport and fate of LA-PAM, the release and fate of the residual acrylamide (AMD) monomer, and the potential impacts on aquatic organisms.
3. Gain a better understanding of how LA-PAM achieves seepage reduction in water delivery canal systems and how various environmental factors affect the ability of LA-PAM to reduce seepage.

The results presented here detail field studies conducted along 17 canal reaches located in the Grand Valley of western Colorado, the Yellowstone River Valley in Montana, and the Lower Arkansas River Valley (LARV) of eastern Colorado, in conjunction with Colorado State University. Results from several laboratory studies directly related to the field application of LA-PAM are also presented. This report complements previous DRI reports that characterized the risk of LA-PAM use in water delivery canals (Young et al., 2007a) and discussed specific LA-PAM application guidelines that reduce environmental exposure while maintaining seepage reduction benefits (Susfalk et al., 2007). In addition, the results of several previous laboratory studies that investigated the mechanisms of seepage reduction, potential impacts on water quality and aquatic species, microbial degradation, and fate and transport of LA-PAM (Young et al., 2007b) are presented.
Benefits of LA-PAM Application

Granular LA-PAM was applied to specific canal reaches at a rate of approximately 11 kg ha\(^{-1}\) (10 lbs ca\(^{-1}\)) based on the average wetted perimeter of each canal. Short-term measurements conducted within 24 hours of each application indicated that LA-PAM reduced seepage rates between 28 and 87 percent in 8 of 11 experiments. For example, the application of LA-PAM along 4 km (2.5 mi) of the LARV Rocky Ford Highline Canal (RFH) in 2006 reduced pretreatment seepage rates of 0.06 m\(^2\) s\(^{-1}\) km\(^{-1}\) by 59 percent. Over a season, the amount of water salvaged from this 4-km reach was estimated to be 1.6 x 10\(^6\) m\(^3\) yr\(^{-1}\) (1,300 acre ft\(^{-1}\) yr\(^{-1}\)). The application of LA-PAM to the entire RFH canal would salvage an estimated 7 x 10\(^6\) to 24 x 10\(^6\) m\(^3\) yr\(^{-1}\) (5,400 to 19,000 acre ft\(^{-1}\) yr\(^{-1}\)) of water for a canal that typically diverts 95 x 10\(^6\) m\(^3\) yr\(^{-1}\) (77,000 acre ft\(^{-1}\) yr\(^{-1}\)).

Based on these field applications, the cost of applying granular LA-PAM ranged from $78 km\(^{-1}\) ($126 mi\(^{-1}\)) for smaller canals to $213 km\(^{-1}\) ($344 mi\(^{-1}\)) for larger canals. On a 10-year annualized basis, this represented between $111 to 202 ha yr\(^{-1}\), or between 0.2 and 3 percent of the total annualized cost of conventional technologies. Stakeholders and/or canal employees that have been trained in the proper field safety and application protocols can apply granular LA-PAM using simple or automated fertilizer spreaders. However, current application devices tend to promote both an uneven distribution and the over-application of LA-PAM. These issues could be eliminated by the development of application devices tailored to this use.

The application of LA-PAM remained effective throughout the remainder of the irrigation season based on repeated measurements conducted at four sites. Seepage reduction benefits were, however, not maintained across the winter season when the canals were dewatered. The need for yearly LA-PAM applications provides greater flexibility in controlling seepage rates compared to traditional methods. For example, some districts may choose to use LA-PAM only during drought conditions when water resources are scarce. Others might delay the use of LA-PAM by several weeks to provide canal seepage to adjacent resources (e.g. wetlands) during the early season when water is more plentiful. Yet others might delay LA-PAM application towards the middle or end of the irrigation season to help stretch the use of dwindling water resources. There is even the potential to halt seepage reduction benefits mid-season by disturbing the LA-PAM induced seal if they are no longer required.

Factors Contributing to a Successful LA-PAM Application

Linear anionic PAM was not always effective at reducing canal seepage, a result of inadequate field conditions necessary to promote flocculation between the polymer and suspended sediment. Granular LA-PAM should only be added to water in canals having a suspended sediment concentration (SSC) of approximately 150 mg L\(^{-1}\) or greater, and a total dissolved concentration (TDS) of approximately 200 mg L\(^{-1}\) or greater. The most common issue was the lack of a sufficient SSC, which could be alleviated by applying LA-PAM during elevated turbidity events such as seasonal snowmelt or after summer thunderstorms. Granular LA-PAM should not be used in canal systems that are chronically devoid of suspended sediment and/or are characterized by low TDS levels that suppress the formation of PAM-sediment flocs. Artificially increasing SSC through dredging the canal bottom or the
addition of canal tailings should be avoided due to the difficulty in timing sediment additions with LA-PAM hydration rates.

The failure to follow these requirements, the application of excessive LA-PAM, and/or the poor choice of application methods and techniques can increase potential environmental exposure through the downstream transport of LA-PAM in the water column. The most critical factor in reducing exposure was the use of an LA-PAM application rate that was related to the level of suspended sediment in the water column and not simply the wetted perimeter, as traditionally done. Results from this study indicate that a nominal application rate of 11 kg ha⁻¹ (10 lbs ac⁻¹) resulted in an over-application of LA-PAM to smaller canals (<2.8 m³ s⁻¹ or < 100 cfs) and potentially resulted in the under-application to larger canals. Smaller canals had less water per unit area, resulting in higher observed concentrations of LA-PAM in these canals. Laboratory studies indicated the formation of PAM-sediment flocs was optimized when LA-PAM concentrations were between 1 to 2 mg L⁻¹ in the water column and that higher polymer concentrations provided little additional benefit. The Clear Zone Index (CZI) was introduced as a diagnostic aid to assess if the mass of LA-PAM exceeded the assimilative ability of suspended sediment levels in the canal. A partially developed CZI was desired, indicating the formation of PAM-sediment flocs and the utilization of added polymer. A fully developed CZI should be avoided, as it indicates exhaustion of suspended sediment and the potential presence of excess LA-PAM that can remain mobile in the water column. It is strongly advised that current application guidelines based on a canal’s wetted perimeter be depreciated in favor of the development of application rates based on the concentration or load of suspended sediment in the water column.

Other environmental factors must also be taken into account to properly manage the application of LA-PAM. The interaction of water temperature and water velocity, for example, will determine how far LA-PAM will travel downstream before it hydrates and reacts with suspended sediment. Laboratory experiments indicated that granular LA-PAM typically hydrates within 24 to 34 minutes. The time needed for flocculation to occur in the field was not entirely temperature dependent, indicating other site-specific factors such as water mixing, water chemistry, suspended sediment concentration and particle size may also play an important role. For the sites studied, LA-PAM traveled between 196 m (643 ft) and 2,149 m (7,050 ft) downstream from the point of application prior to flocculation and development of a clear zone. Therefore, granular LA-PAM must be applied at greater distances upstream of the target reach under conditions of slow hydration rates and fast water velocities, such as typical of snowmelt-fed canals during the early water season.

**Risks of LA-PAM Application**

The application of LA-PAM to unlined water delivery canals carries several potential risks related to the application and release of the polymer and the residual AMD monomer that remains occluded in the polymer. When LA-PAM was applied in excess of ambient suspended sediment conditions, LA-PAM concentrations were found to exceed 1 mg L⁻¹ for up to four to nine hours depending on the time it took to physically apply the polymer. Excess LA-PAM that remained in the water column could travel significant distances downstream where it could be inadvertently used by unsuspecting stakeholders, such as being diverted to farms for use on crops, or be consumed by livestock, for example. Excess polymer can also negatively impact the aquatic community. In canal systems, the response of benthic macroinvertebrates (BMIs) to LA-PAM was relatively minor because these BMI
communities were tolerant of the very harsh conditions present in these types of systems. However, natural surface water systems are comprised of more sensitive BMI species that were found to respond negatively to the presence of LA-PAM. The primary approach for mitigating these risks is to reduce or eliminate the transport of polymer downstream of the treatment zone by applying LA-PAM in a manner that assures it is quickly removed from the water column. This requires proper water chemistry, use consistent with guidelines developed for LA-PAM application to canals, and the application of LA-PAM at a rate that will not fully deplete the load of suspended sediment carried by the canal. These requirements may rule out or delay the use of LA-PAM until sufficient sediment levels are present, or even prevent its use if there is a possibility that natural surface waters may be impacted.

The most likely human health risk was to persons applying LA-PAM, either through inhalation, accidental eye contact, or from slipping on the slick polymer as it hydrated. These risks were significantly reduced through the proper use of protective gear and application guidelines (Susfalk et al., 2007). The incidental release of the AMD monomer, a cumulative neurotoxin and a suspected human carcinogen, also presents potential human health and environmental risks. Unlike the polymer, AMD is a small, mobile molecule that can enter groundwater. The potential risks of AMD release associated with LA-PAM applications to canal systems were more fully discussed in Young et al. (2007a). Acrylamide concentrations in canal water were observed to be orders of magnitude below the chronic levels needed to impact human health. In addition, there was evidence that elevated AMD and LA-PAM concentrations were linked, suggesting that the methods discussed above to eliminate excess polymer addition should also reduce the potential for elevated monomer concentrations. The successful formation of a PAM-sediment-induced seal will also decrease the likelihood of AMD entry into groundwater through the reduction of the seepage rate. Acrylamide entering groundwater was found to be diluted by both canal water and groundwater, a dilution of up to four orders of magnitude at one site. Acrylamide released into the environment was found to be susceptible to microbial degradation, with an estimated half-life of 30 to 42 hours. Finally, conservative transport models indicated that AMD concentrations 10 times greater than actually measured in canal water would be undetectable within 25 m of the canal due to microbial degradation and dilution processes. Therefore, the contamination of groundwater by AMD associated with the application of LA-PAM to water delivery canals using the methods of Susfalk et al. (2007) was considered to be very unlikely.

The successful use of granular polyacrylamide includes: 1) the selection of the proper type of LA-PAM for use in canals; 2) determination if the proper water chemistry and suspended sediment concentrations exist; 3) an application plan that minimizes worker contact and accounts for environmental impacts on hydration time; and 4) an assessment of the potential risks including potential impacts on downstream users and receiving waters. Stakeholders and/or agencies responsible for the use of granular LA-PAM must assess if potential site-specific risks outweigh the benefits of using LA-PAM for water conservation purposes.
ACKNOWLEDGEMENTS

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# ABBREVIATIONS

<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Description</th>
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<tr>
<td>ADCP</td>
<td>acoustic doppler current profiler</td>
</tr>
<tr>
<td>ADV</td>
<td>acoustic doppler velocimeter</td>
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<tr>
<td>AMD</td>
<td>acrylamide monomer</td>
</tr>
<tr>
<td>ANOVA</td>
<td>Analysis of variance</td>
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<tr>
<td>ANSI</td>
<td>American National Standards Institute</td>
</tr>
<tr>
<td>ASTM</td>
<td>American Society for Testing and Materials</td>
</tr>
<tr>
<td>BMI</td>
<td>benthic macroinvertebrate</td>
</tr>
<tr>
<td>CAT</td>
<td>Catlin Canal</td>
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<tr>
<td>CCA</td>
<td>canonical correspondence analysis</td>
</tr>
<tr>
<td>CDWR</td>
<td>Colorado Department of Water Resources</td>
</tr>
<tr>
<td>CSU</td>
<td>Colorado State University</td>
</tr>
<tr>
<td>DCA</td>
<td>detrended correspondence analysis</td>
</tr>
<tr>
<td>DI</td>
<td>de-ionized</td>
</tr>
<tr>
<td>DOC</td>
<td>dissolved organic carbon</td>
</tr>
<tr>
<td>DRI</td>
<td>Desert Research Institute</td>
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<tr>
<td>FL</td>
<td>Fort Lyon Canal</td>
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<tr>
<td>GVIC</td>
<td>Grand Valley Irrigation Company</td>
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<tr>
<td>HBI</td>
<td>Hilsenhoff biotic index</td>
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<tr>
<td>HID</td>
<td>Huntley Irrigation District</td>
</tr>
<tr>
<td>ID</td>
<td>Irrigation District</td>
</tr>
<tr>
<td>KC</td>
<td>Kannah Creek Ditch #2</td>
</tr>
<tr>
<td>LAM</td>
<td>Lamar Canal Company</td>
</tr>
<tr>
<td>LARV</td>
<td>Lower Arkansas River Valley</td>
</tr>
<tr>
<td>LA-PAM</td>
<td>linear anionic polyacrylamide</td>
</tr>
<tr>
<td>LPSA</td>
<td>laser particle size analysis</td>
</tr>
<tr>
<td>MD</td>
<td>Minnesota Ditch</td>
</tr>
<tr>
<td>NOAEL</td>
<td>No-observed-adverse-affect-level</td>
</tr>
<tr>
<td>NRCS</td>
<td>Natural Resources Conservation Service</td>
</tr>
<tr>
<td>NSF</td>
<td>National Science Foundation</td>
</tr>
<tr>
<td>NTU</td>
<td>nephelometric turbidity units</td>
</tr>
<tr>
<td>PAM</td>
<td>polyacrylamide</td>
</tr>
<tr>
<td>RFH</td>
<td>Rocky Ford Highline Canal</td>
</tr>
<tr>
<td>SAR</td>
<td>sodium adsorption ratio</td>
</tr>
<tr>
<td>SCL</td>
<td>Soil Characterization Laboratory</td>
</tr>
<tr>
<td>SD</td>
<td>Smith Ditch #7</td>
</tr>
<tr>
<td>SIP</td>
<td>slope of inflection point</td>
</tr>
<tr>
<td>SSC</td>
<td>suspended sediments concentration</td>
</tr>
<tr>
<td>TDS</td>
<td>total dissolved solids</td>
</tr>
<tr>
<td>TSS</td>
<td>total suspended solids</td>
</tr>
<tr>
<td>UNC</td>
<td>Uncompaghre Irrigation District</td>
</tr>
<tr>
<td>UNR</td>
<td>University of Nevada, Reno</td>
</tr>
<tr>
<td>USBR</td>
<td>United States Bureau of Reclamation</td>
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<tr>
<td>USDA</td>
<td>United States Department of Agriculture</td>
</tr>
<tr>
<td>USEPA</td>
<td>United States Environmental Protection Agency</td>
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<tr>
<td>USGS</td>
<td>United States Geological Survey</td>
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1.0 INTRODUCTION

The loss of water during transport through unlined water delivery and irrigation canals can be significant, with as much as 50 percent lost to seepage through the sides and bottoms of the canals (USGS, 1990). Seepage losses are difficult to quantify as they are highly variable in space and time. Spatial variability is attributed in part to local soils and geology, groundwater characteristics, and sediment concentrations, while temporal variability is due to management of canal reaches during the agricultural season, and the influence of seasonal and localized hydrologic events. Although the complexity of seepage makes regional estimates difficult to quantify, there is little doubt that seepage does occur on a large scale, and that it has negative impacts in some areas. In 1978, it was estimated that 37 percent of the 1.1 million ac-ft yr$^{-1}$ of water in 30,000 mi of unlined canals in the western United States was lost to seepage (USDOI, 1978). United States Bureau of Reclamation (USBR) water projects encompass 16,000 mi of off-farm water delivery canals (USDOI, 1948), of which an estimated 14,000 mi are unlined. As a result, there is strong interest in the use of cost-effective, widely applicable technologies to reduce the impacts of canal seepage and bring benefits to drought-stricken areas of the U.S.

Recently, linear anionic polyacrylamide (LA-PAM) has been identified for its ability to reduce seepage from unlined canals. Polyacrylamides consist of a broad family of polymers that have been extensively used in food packaging, paper manufacturing, and as flocculants in wastewater treatment. Individual polymers within the polyacrylamide family have various physical and chemical properties, as noted by the diversity of their usage. Over the past decade, LA-PAM has been used to reduce erosion and sediment transport from crop fields under furrow irrigation (Wallace and Wallace, 1986; Lentz and Sojka, 1994; Lentz et al., 2001; Al-Abed et al., 2003) and on construction sites (Soupir et al., 2004; Hayes et al., 2005; Orts et al., 2007).

The primary benefit of using LA-PAM in irrigation canals is to cost-effectively reduce seepage losses through the bottom and side walls of unlined canals. Traditional seepage-abatement technologies such as compacted earth, reinforced or unreinforced concrete, and buried geomembranes are typically used in situations where seepage rates are high and the projected water savings offset their costly construction and maintenance costs. In contrast, LA-PAM is relatively inexpensive. Granular LA-PAM is easy to apply and can be targeted to specific canal reaches known to have high seepage rates. Water savings realized from LA-PAM could result in lower amounts of water being diverted into the canals and could also result in the extension of irrigation water supplies and the duration of the irrigation season. There is conflicting empirical evidence of the longevity of a single LA-PAM application, but yearly or more frequent applications do not diminish its cost-effectiveness. This “short-term” seepage reduction of LA-PAM, relative to the more permanent nature of traditional technologies, is a benefit, as it could be applied selectively during drought water years, and allowed to elapse if water savings are no longer needed.

There are secondary benefits to controlling seepage with LA-PAM. First, some canal operators have found that their canal system is much more responsive after LA-PAM application, providing them the ability to address downstream water needs faster (P. Bertrand, Grand Valley Irrigation Company, personal communication, 2005). More importantly, controlling seepage reduces water flux to local groundwater systems, especially
in areas characterized by already high groundwater levels and/or underlying geology high in soluble salts. In the Lower Arkansas River Valley (LARV) of eastern Colorado, for instance, the 1,000 miles of canals that have made a productive agricultural economy possible have also contributed to decreased depths to groundwater. One detrimental result of shallow water tables is a 10 to 15 percent drop in crop yields due to waterlogged crops and increased salt concentrations through evaporative concentration (Burkhalter and Gates, 2005; Gates et al., 2006). Another is that built-up water tables drive increased groundwater return flows that dissolve and pick up salts and metals from marine shales and residuum as they make their way back to the river system (Mueller Price and Gates, 2008). Finally, upflux from shallow water tables that extend under adjacent naturally vegetated and fallow land results in the substantial loss of water to nonbeneficial consumptive use.

Similar processes occur in the Grand Valley of western Colorado, where a reduction in the rate of deep percolation of canal seepage is a higher priority than seepage reduction for water conservation. Like in the LARV, as canal seepage-sourced groundwater moves toward the Colorado River, it dissolves soluble salts and metals from the underlying strata. The economic damage throughout the Colorado River Basin from highly saline waters has been estimated at $330 million per year and has resulted in the creation of the Grand Valley Unit under Title II of the Salinity Control Act, which is tasked to reduce salt added to the Colorado River by 580,000 ton yr\(^{-1}\) (USDOI, 2003).

The potential risks involved with the application of LA-PAM include worker exposure during application and the fate and transport of both PAM and the residual acrylamide monomer (AMD) in the environment. During LA-PAM application, worker exposure can be controlled with the proper safety equipment. The release of PAM and AMD into the environment, however, could have adverse impacts on the aquatic community, especially in areas where LA-PAM-treated canal waters discharge into natural receiving waters. Young et al. (2007a) have found that the application of LA-PAM to water delivery canals poses minimal human health risks. Risks can be further reduced when application guidelines that minimize environmental exposure are followed (Susfalk et al., 2007).

In 2005, the Desert Research Institute (DRI), in collaboration with the USBR, initiated laboratory and field studies to assess the benefits and risks of LA-PAM use as a method to reduce seepage and conserve water in unlined water delivery canals. The scope of these studies was broad, as the performance and environmental impacts of an individual polymer could not be predicted based solely on its chemical or physical properties. A Peer Review Panel (PRP) comprised of outside scientists from university, government, industry, and stakeholder groups was established to assist with the development of a risk characterization report (Young et al., 2007a) and to provide guidance to the researchers involved with the study. The PRP initiated a series of questions that later developed into a set of seven research questions that guided the work presented here:

1. What are the ecological and human health risks of the use of PAM and AMD in unlined canals for seepage control?
2. Does PAM (the polymer) degrade to the monomer, acrylamide (AMD)? If so, does the amount present a significant risk for contamination of surface water or groundwater?
3. What is the relative significance of residual AMD in the original polymer versus AMD as a PAM degradation product (if it is generated)? Are there other potential or known degradation products of PAM that are of toxicological concern?

4. What is the fate (including biodegradation) and transport of AMD (and/or PAM, and product components) in surface water, soil, and groundwater systems? What data gaps exist specific to this application?

5. How do field application practices (e.g., application of PAM to dry soil versus water in a flowing canal) affect the risk in using PAM? What field practices can be used to reduce risks associated with PAM application?

6. If residual PAM is released into receiving water, what are the ecological risks and issues associated with PAM in surface water (e.g., armoring, channel morphology, bioaccumulation, etc.)?

7. Are there any other issues regarding the human and ecological risk in using PAM that should be considered?

Field, laboratory, and numerical studies were initiated to address the above questions that emphasized the fate and transport of LA-PAM and AMD in the canal environment and the characterization of ecological risk from LA-PAM applications. The objectives of laboratory studies were to: 1) characterize the potential environmental impacts of LA-PAM use; 2) investigate the mechanisms, impacts on water quality, aquatic species, microbial degradation, and fate and transport of LA-PAM (Young et al., 2007b; Susfalk et al., in preparation); and 3) provide suggested guidelines for the use of LA-PAM in canal systems that reduce environmental exposure while maintaining seepage reduction benefits (Susfalk et al., 2007). Specific objectives of the field-based studies presented in this report were to: 1) describe how PAM reduces seepage in unlined canal systems and the factors that affect its efficacy; 2) quantify the potential seepage reduction benefits, water savings, and typical application cost; and 3) address potential risks of LA-PAM application, including AMD release, PAM release, and potential impacts on aquatic organisms.
2.0 LA-PAM APPLICATION METHODS AND SITES

Research was conducted within active water delivery canals during three field seasons, from 2005 through 2007, with different objectives each year. A list of these canals and their associated acronyms used in this report are presented in Table 2-1. Studies conducted in 2005 were generally in low-volume canals (<0.85 m$^3$ s$^{-1}$, <30 cfs) to gain experience in the use, application, and measurement of LA-PAM and AMD in canal systems. The most important experiment during this year was at SD, where a spatially and temporally intense dataset on LA-PAM uses and its impact on water quality and the aquatic community was developed. The results from that study solidified the LA-PAM application and monitoring techniques used in this study and formed the foundation that was used in all subsequent studies.

In 2006, the objective was to assess the short-term change in seepage following the application of LA-PAM at a large number of sites in larger-volume canals (<2.8 m$^3$ s$^{-1}$, >100 cfs). As there was a general lack of historical seepage estimates, site visits were necessary to determine if a potential target reach was suitable for seepage measurement. Several factors had to be considered, including 1) a measurable volume of seepage loss; 2) the stability of inflows impacted either by precipitation or canal management practices; and 3) the stability and/or control of turn-outs feeding individual stakeholders. To complicate this, seepage rates can vary during the irrigation season, flows are highly responsive to regional snowmelt and precipitation patterns, and canals are actively managed to deliver water to their stakeholders, sometimes in unpredictable ways. Therefore, the most efficient and cost-effective procedure was to conduct the entire experiment during one site visit. Site visits were arranged within one to two weeks notice to take advantage of favorable on-the-ground conditions. Nine experiments were conducted during 2007, each lasting approximately one week. Experiments included planning, reach assessment, pre-treatment seepage measurements, LA-PAM application, subsequent seepage measurements, sample collection lasting up to 36 hours, sample processing, and the transport of samples back to the laboratory. Field conditions sometimes precluded the measurement of seepage either prior to or after LA-PAM application. The focus of these experiments was on the concentration and downstream transport of LA-PAM in canal water.

Post-application seepage measurements were repeated throughout the remainder of the 2006 irrigation season in several canal reaches to assess the longevity of seepage reduction. However, the lack of suitable control reaches and a long-term seepage record at these sites prior to LA-PAM application made it difficult to assess the season-long impact that LA-PAM had relative to native, untreated canals. The use of control sections was not viable in most sites due to the inherent differences in seepage and canal management between adjacent reaches of the same canal. This was exemplified by the RFH, where the canal reach designated as a control was found to have six times greater seepage than the treated reach.

The large number of sites visited in 2006 provided essential background information on water chemistry and native seepage rates useful in gaining an understanding of the environmental conditions required for the successful application of LA-PAM. It also provided the opportunity to screen these sites and choose the best sites for more in-depth studies the following year. The objective of the 2007 studies was to characterize native
(non-treated) seepage losses over the majority of the irrigation season at a few sites to better understand the natural variability in seepage rates and to provide a baseline with which to contrast results from LA-PAM treatments.

Table 2-1. Methods used for the application and environmental monitoring of LA-PAM, by site.

<table>
<thead>
<tr>
<th>Year</th>
<th>Canal</th>
<th>Application Type</th>
<th>Spreader</th>
<th>Obs.</th>
<th>AMD</th>
<th>Seepage Before</th>
<th>Seepage After</th>
<th>Suspended Sediments</th>
</tr>
</thead>
<tbody>
<tr>
<td>2006</td>
<td>CAT-1 Catlin Canal</td>
<td>M BH 1 S</td>
<td>S</td>
<td>S</td>
<td>M</td>
<td>G</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2007</td>
<td>CAT-2 Catlin Canal</td>
<td>M BH 1 S, T M M</td>
<td>G, TA</td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>2006</td>
<td>GVIC-1 Grand Valley Irrigation Co.</td>
<td>D -- M S -- -- --</td>
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<tr>
<td>2006</td>
<td>HID-1 Huntley ID</td>
<td>S HL M --</td>
<td>S</td>
<td>S</td>
<td>G, TP</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006</td>
<td>HID-2 Huntley ID</td>
<td>S HL M --</td>
<td>S</td>
<td>S</td>
<td>G, TP</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2005</td>
<td>KC-1 Kannah Creek Ditch #2</td>
<td>M HS M S S S G</td>
<td></td>
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<td></td>
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<td></td>
</tr>
<tr>
<td>2006</td>
<td>KC-2 Kannah Creek Ditch #2</td>
<td>M HS M -- S S G</td>
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<td></td>
</tr>
<tr>
<td>2006</td>
<td>LAM-1 Lamar ID</td>
<td>M HS 1 --</td>
<td>S</td>
<td>S</td>
<td>G</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006</td>
<td>LAM-2 Lamar ID</td>
<td>M BH 1 S S S G</td>
<td></td>
<td></td>
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<tr>
<td>2006</td>
<td>MD Minnesota Ditch</td>
<td>M HS M -- S S G</td>
<td></td>
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</tr>
<tr>
<td>2006</td>
<td>RFH-1 Rocky Ford Highline</td>
<td>M BA 1 --</td>
<td>S</td>
<td>M</td>
<td>G, TA</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006</td>
<td>RFH-2 Rocky Ford Highline</td>
<td>M BA S</td>
<td>S</td>
<td>M</td>
<td>TA</td>
<td></td>
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<tr>
<td>2007</td>
<td>RFH-3 Rocky Ford Highline</td>
<td>M BA -- -- -- -- --</td>
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<tr>
<td>2007</td>
<td>RFH-4 Rocky Ford Highline</td>
<td>S HS M -- -- -- TP</td>
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<tr>
<td>2007</td>
<td>RFH-5 Rocky Ford Highline</td>
<td>-- -- -- -- -- M -- TA</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2005</td>
<td>SD Smith Ditch #7</td>
<td>M HS M S S S G, TA</td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006</td>
<td>UNC Uncompaghre ID</td>
<td>M BA 1 -- S S G, TA</td>
<td></td>
<td></td>
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</tr>
</tbody>
</table>

LA-PAM was applied at a stationary point (S), moving upstream (M), or as emulsified LA-PAM applied to a dry canal (D). The LA-PAM was applied by walking down or alongside the canal using a small (HS) or large (HL) hand spreader, or by boat using a small hand spreader (BH) or a large hopper spreader (BA). Downstream LA-PAM concentrations were monitored at either one (1) or multiple (M) locations. Water samples from within the canal were collected for AMD analysis (S), and as part of a dye-tracer study (T) conducted to assess potential groundwater transport of AMD. Seepage was measured before and after LA-PAM application for a short time period (S) ranging from one to 14 days, or for medium time period (M) ranging from one to six months. Suspended sediment levels were assessed in the canal using grab samples (G) analyzed for suspended sediment concentration and/or with turbidity measured by an automated meter (TA) placed within the canal or by grab samples analyzed by a portable meter (TP). Details regarding each site are presented in Section 2.3, and details regarding the specific methodology are presented in Section 2.1 to 2.3.

2.1 Polyacrylamide

Acrylamide polymers, or polyacrylamides (PAMs), are a family of water-soluble molecules that have a range of charges, charge densities, and molecular weights. Polyacrylamides were developed in the late 1800s and have been commercially available since the 1950s (Kurenkov, 1997). They were initially used as floculants for the treatment of acid-leached uranium and as a dry strength agent in paper manufacturing. Polyacrylamides can carry a variety of charges including non-ionic, cationic, and anionic. Non-ionic cross-linked PAM has been used in applications such as the clarification of potable water, the absorbent in some baby diapers, and more recently as a subdermal filler for aesthetic facial surgery and in gel electrophoresis. Linear PAM molecules possess either a cationic or anionic charge and have been traditionally used as a thickener and suspending agent for the separation and clarification of liquid-solid phases, such as in the treatment of wastewater. Linear anionic polyacrylamides (LA-PAM) have typically been used in environmental
applications because they are more effective at low concentrations and possesses a lower risk of toxicological impacts than cationic forms (Sojka et al., 2007).

The molecular weight of linear PAMs varies from about 1.0 to about 20 million grams per mole (g mol\(^{-1}\)), with cationic forms usually at the lower end and anionic forms usually at the higher end of this range. The charge density, or extent of electrically charged sites along the PAM chain, is variable, but most effective LA-PAMs have a charge density of 20 to 30 percent.

Physically, PAM can be purchased as a liquid dispersion, a liquid emulsion, and as a coarse or fine-grained solid. The studies presented here were limited to only the coarse form of granular LA-PAM, specifically applied in granular form to a water-filled canal. This was considered to be the most common application technique and of most value to the greatest number of canal managers and stakeholders. Liquid LA-PAM forms (dispersed or emulsion) are typically more expensive and require a significant investment in equipment or the need to hire an outside contractor specializing in liquid applications to properly mix and distribute PAM to the canal. However, the selection of liquid forms may be more logical in several situations, such as the need to apply LA-PAM to a dewatered canal.

### 2.2 LA-PAM and Application Methods

The LA-PAM used during these studies conformed to U.S. Department of Agriculture (USDA) interim specifications for use in on-farm irrigation canals (NRCS, 2005). More specifically, the manufacturer certified that the LA-PAM 1) complied with the NSF International/American National Standards Institute (ANSI) Health Effect Standard (NSF/ANSI-60); 2) was linear (not deliberately branched) with an anionic charge density of 30 percent plus or minus 5 percent; 3) had a molecular weight of at least 12 million g (Mg) mole\(^{-1}\); and 4) contained no more than 0.05 percent acrylamide monomer, by weight. The LA-PAM polymer used for field experiments at HID, KC, MD, SD, UNC, and in the laboratory was Tack Dry, distributed by Precision Polymer Corporation of Greeley, Colorado. This LA-PAM had a molecular weight of approximately 18 Mg mole\(^{-1}\) and was found to contain 0.01 percent of AMD (Jim Woodrow, University of Nevada, Reno, personal communication, July 2006). For the remaining field applications, the LA-PAM polymer used was Stokopam distributed by JT Water Management, LLC of Parker, Colorado. Both LA-PAM products were assumed to contain a 92 percent active ingredient (active LA-PAM) with 4 percent percent moisture for purposes of calculating application rates.

The Phase II Rule National Primary Drinking Water Regulations issued by the U.S. Environmental Protection Agency (USEPA) (40 CFR §141.111) asserts an acrylamide polymer maximum use level of 1.0 mg L\(^{-1}\) and an AMD concentration of 0.05 percent in the polymer, or equivalent, for a carryover of not more than 0.5 \(\mu\)g L\(^{-1}\) of AMD into the finished water. Because canal water could possibly be used directly or indirectly for human consumption, we sought to apply LA-PAM to achieve a canal water concentration of less than 1.0 mg L\(^{-1}\).

During field applications in 2005 and early 2006, application methods and monitoring protocols were being refined. For simplicity, the application methods presented here describe the final methods that were developed during this timeframe. Significant, site-specific deviations to these general procedures are addressed in Section 2.3. Applications were
generally conducted by moving the point of LA-PAM application upstream while continuously dispersing granular LA-PAM onto the canal water surface by walking or by boat (Table 2-1). In addition, three experiments were conducted by applying LA-PAM at a stationary application point to better assess the interaction of LA-PAM and suspended sediment. In all instances, technicians observed good industrial hygiene safety practices, which included the use of protective eyewear, dust masks, and gloves. Excess LA-PAM resulting from weighing and transfer functions was immediately cleaned up, because, once exposed to water, LA-PAM is extremely slippery and difficult to remove from surfaces.

Application rates of LA-PAM were determined using the units of kilograms per canal hectare (kg ha\(^{-1}\)), or pounds per canal acre (lbs ac\(^{-1}\)), which are both based on wetted perimeter. Unless otherwise noted, application rates reported here are “effective” rates based on active LA-PAM. Application rates of 11.2 to 16.8 kg of LA-PAM per wetted canal ha (10 to 15 lbs ac\(^{-1}\)) were targeted for each of the applications in 2006 and 2007. Prior to starting an application, the total weight of LA-PAM for application was calculated by multiplying the target application rate by the estimated wetted perimeter area of the study reach. Enough LA-PAM for 0.16 km (0.1 mi) of the study reach was then determined, and the appropriate amount of LA-PAM was weighed and placed in individual, sealing plastic bags. During the application process, a pace-setting truck was driven ahead of the application technicians in the canal and stopped to resupply the technicians with LA-PAM. When canal conditions were not appropriate—for example, if turbidity or seepage were low—LA-PAM was not applied.

2.2.1 Motorized Boat Application

The benefits of using a boat for LA-PAM application are realized in larger canals. Benefits include time efficiency, well-balanced concentration, and increased hydration of LA-PAM (better mixing caused by the turbulence from the boat prop). Either several hand (manual) spreaders or an automated spreader was used to apply LA-PAM. Modification to the automated spreader (Herd, Logansport, IN) allowed LA-PAM to be applied over a window of approximately 180 degrees in front of the boat, covering the full canal width. Thus, only one application technician and a boat driver were required. Increased boat speed and higher release height, however, resulted in some reduced control over landing area of LA-PAM (some landed in the boat). The automated spreader was used only when it would not spread LA-PAM onto the canal bank (“HS” in Table 2-1, Figure 2-1).

When the automated spreader was not used, LA-PAM was applied with two small hand spreaders (Scotts, Marysville, OH) from each side of the boat (“BH” in Table 2-1, Figure 2-2). Application speeds ranged from 1.4 to 1.6 km hr\(^{-1}\) (0.9 to 1.0 mi hr\(^{-1}\)), when using a hand spreader, to 2.1 to 4.8 km hr\(^{-1}\) (1.3 to 3.0 mi hr\(^{-1}\)), when using the automated spreader. While the automated spreader may be more time-efficient, slower application rates may ultimately result in lower concentrations of LA-PAM in canal water.
2.2.2 Walking Application

When boat use was not feasible in smaller canals, two technicians with hand spreaders walked the canal, traveling in the upstream direction. If application was done from the bank of the canal, three spreader turns (approximately 28 g of LA-PAM) were applied at 30-m (100-ft) intervals. If access to the canal bank was not possible, the technicians applied LA-PAM from the center of the canal prism. A larger, chest-mounted hand spreader (Earthway, Bristol, IN) was used (“HL” in Table 2-1) for the stationary studies in Montana. The larger hopper and handle made LA-PAM application easier. Application times, when walking within the canal prism, were about 0.5 km hr$^{-1}$ (0.3 mi hr$^{-1}$), and increased to 0.6 to 1.3 km hr$^{-1}$ (0.4 to 0.8 mi hr$^{-1}$) when walking on the banks of the canal (Figure 2-3).
Figure 2-3. LA-PAM application by walking the RFH (top) and SD (bottom). At SD, the backhoe was used to locally elevate suspended sediment in the canal.

2.2.3 Stationary Application

Besides mobile application methods, LA-PAM was also applied continuously from a stationary (bridge) location, for a period of 90 to 120 minutes (Figure 2-4). This application method was used to observe downstream turbidity changes in response to the formation of PAM-sediment flocs. Three locations were studied in this manner, two canal reaches of HID in 2006 and one canal reach along RFH in 2007. A time-based sampling approach was used at HID, whereas sampling was concentrated to times when visually observable changes in turbidity were noted at RFH.

Control samples were taken from the center of the upstream side of the bridge where LA-PAM was applied. Depth-integrated samples were collected from either the bank of the canal or from the center of a bridge, if present. Water samples were collected with a US DH-81 (depth-integrating suspended sediment sampler), and subsampled into a 50-mL
centrifuge tube for turbidity analysis and a 500-mL bottle for suspended solids concentration (SSC) analysis. A second water sample was collected for LA-PAM analysis following standard procedures (Section 3.4). Turbidity samples were stored at ambient temperature and analyzed with a 2020e turbidimeter (LaMotte, Chestertown, MD) within 24 hours. Samples for SSC analysis were shipped back to DRI for analysis. A total of 50 SSC, 62 LA-PAM, and 190 turbidity samples were collected during both experiments at HID. During the RFH application, 1 SSC, 95 LA-PAM, and 95 turbidity samples were collected. Discharge was quantified using a RDI-Teledyne StreamPro acoustic Doppler current profiler (ACDP) following standard methods presented in Section 3.1.2.2.

![Figure 2-4. Application of LA-PAM off a bridge at HID.](image)

### 2.3 Site Selection

Criteria used to select study reaches included: 1) magnitude of seepage loss; 2) controllable or stable inflows; 3) minimum number of noncontrollable turn-outs or returns; 4) length of canal; 5) presence of an upstream control reach; 6) presence of background data; and 7) ability to collect downstream water chemistry samples (Table 2-2). The selection of the control and treatment reaches attempted to balance these criteria. A total of 17 LA-PAM application experiments were conducted from 2005 to 2007. During 2005 and early 2006, field results for LA-PAM effectiveness and potential water quality issues were needed as quickly as possible to aid USBR decision-making processes on LA-PAM usage. As a result, a considerable number of LA-PAM applications were conducted, but many without control reaches or background conditions that were monitored longer than several days. This approach caused difficulties when evaluating seasonal seepage, but provided excellent data on both short-term seepage loss and LA-PAM release into the water column. Perhaps more importantly, it allowed for refinement of LA-PAM application procedures. Later studies, during 2007, focused on obtaining significant background data and gaining a better understanding of naturally occurring changes in seepage.
2.4 LA-PAM Application

2.4.1 Catlin Canal, Rocky Ford, CO

The Catlin Canal (CAT) is an agricultural water delivery canal diverted from the Arkansas River near Manzanola in southeastern Colorado. Two applications of LA-PAM were conducted along the same reach of CAT. The first application (CAT-1) was conducted on June 3, 2006, at 3:30 pm and lasted for 2.5 hours. The study reach was located south of Rocky Ford (Figure 2-5) and extended 3.9 km (2.4 mi) from site 201 downstream to site 202. Granular LA-PAM was applied to the water from a motorized boat using two hand spreaders, following the method described in Section 2.1.1. All diversions from the canal reach were shut off two days before and after the application to ensure that the treated water remained within the canal, and to reduce seepage measurement errors.

A second application (CAT-2) was conducted on August 7, 2007, using the same methodology. The application began at the Otero County Road 21 bridge (0.5 km downstream of site 202) and extended 4.8 km (3.0 mi) upstream to the siphon at Road Y.00 (0.3 km upstream of site 201) and lasted approximately 2.75 hours. The application reach was extended 0.3 km (0.2 mi) upstream of site 201, so that LA-PAM had more time to hydrate prior to reaching the upstream boundary of the control volume. Few obstructions, including low-lying bridges, flumes, and fences, were located along this reach of the canal, so boat travel was relatively easy. A total of 66.6 kg (147 lbs) of LA-PAM were applied to the reach with 2.0 kg (4.45 lbs) added to the water every 0.16 km (0.1 mi). Toward the upstream end of the study reach, the canal widens for roughly 1.0 km (0.6 mi). To apply approximately the same rate of LA-PAM, 2.9 kg of LA-PAM per 0.16 km (6.7 lbs per 0.1 mi) were added to this wider section. The total application rate was about 18.3 kg ha\(^{-1}\) (16.3 lbs ac\(^{-1}\)).

To determine the long-term effectiveness of LA-PAM treatment, seepage measurements were conducted in 2007 using differential surface water methodology along the same reach over a span of 20 weeks prior to another LA-PAM application. Pre-treatment seepage rates were significantly greater than post-treatment rates measured in 2006, indicating that the 2006 application had lost its effectiveness between water years. On August 7, 2007, LA-PAM was applied using a boat and two hand spreaders. A total of 46 kg (101 lbs) of LA-PAM were applied over two hours at a rate of 12.1 kg ha\(^{-1}\) (10.8 lbs ac\(^{-1}\)) or 7.5 kg km\(^{-1}\) (26.7 lbs mi\(^{-1}\)).
### Table 2-2. Summary of LA-PAM application locations. Sites refer to locations presented in Table 2-1. Numbers present in the code refer to application experiment. Some experimental reaches were partitioned into two or more subsets and are reported on separate lines below. Coordinate datum is WGS-84.

<table>
<thead>
<tr>
<th>Code</th>
<th>Beginning Latitude</th>
<th>Beginning Longitude</th>
<th>End Latitude</th>
<th>End Longitude</th>
<th>Wetted Perimeter, m (ft)</th>
<th>Typical seepage, m³ s⁻¹ km⁻¹ (ft³ s⁻¹ mi⁻¹)</th>
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</thead>
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<td>CAT-1</td>
<td>37.952828</td>
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<td>37.972989</td>
<td>-103.695466</td>
<td>8.5 (28)</td>
<td>0.05 (2.59)</td>
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<td>39.228611</td>
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<td>--</td>
</tr>
<tr>
<td>HID-1</td>
<td>45.9268592</td>
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<td>45.9343765</td>
<td>-108.185009</td>
<td>16.03 (52.6)</td>
<td>0.14 (7.98)</td>
</tr>
<tr>
<td>HID-2</td>
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<td>-108.2796908</td>
<td>45.9218172</td>
<td>-108.2429837</td>
<td>16.55 (54.3)</td>
<td>0.08 (4.61)</td>
</tr>
<tr>
<td>KC-1</td>
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<td>38.96121667</td>
<td>-108.4202944</td>
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<td>--</td>
</tr>
<tr>
<td>KC-2</td>
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<td>38.9761</td>
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<td>--</td>
</tr>
<tr>
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<td>-102.5310298</td>
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<tr>
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<td>5.94 (19.5)</td>
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<tr>
<td>LAM-2</td>
<td>38.0817331</td>
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<td>-102.4216285</td>
<td>6.33 (20.8)</td>
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<tr>
<td>LAM-2</td>
<td>38.0683866</td>
<td>-102.4088998</td>
<td>38.0641353</td>
<td>-102.3708587</td>
<td>5.82 (19.1)</td>
<td>0.01 (0.62)</td>
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<td>RFH-1</td>
<td>38.1849430</td>
<td>-104.1772963</td>
<td>38.08029192</td>
<td>11.27 (37)</td>
<td>0.04 (2.3)</td>
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<td>-104.01161873</td>
<td>9.81 (32.2)</td>
<td>0.02 (0.95)</td>
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<td>38.18494304</td>
<td>-104.17729629</td>
<td>13.1 (43)</td>
<td>0.02 (1.19)</td>
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<tr>
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<td>-104.1772963</td>
<td>38.18343845</td>
<td>-104.17090592</td>
<td>11.49 (37.7)</td>
<td>0.05 (3.12)</td>
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<td>--</td>
</tr>
<tr>
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<td>38.18494304</td>
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<td>--</td>
</tr>
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<td>38.92898333</td>
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<td>--</td>
</tr>
<tr>
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<td>--</td>
</tr>
</tbody>
</table>
2.4.2 Grand Valley Irrigation Company, Grand Junction, CO

Since 1917, water has been diverted into the Government Highline Canal at the Grand Valley Project Diversion Dam, about 37 km (23 mi) northeast of Grand Junction. The canal is on the west and north sides of the river and extends from the Grand Valley Project Diversion Dam south and west a distance of 89 km (55 mi) (http://www.usbr.gov/dataweb/html/grandvalley.html).

Approximately 1,075 kg km\(^{-2}\) (9.6 lbs ac\(^{-1}\)) of Tack Liquid LA-PAM (Precision Polymer, Greeley, CO) was applied to 144 km (90 mi) of the Grand Valley Irrigation Company (GVIC) canal system (Figure 2-6) between March 27 and 30, 2006, after the canal had been dry all winter. Approximately 8 km (5 mi) of the Main Canal were treated with a partially hydrated LA-PAM solution. The entire Upper and Lower Mainline canals were treated with 379 L (100 gal) water mixed with 11 L (3 gal) of 40 percent PAM emulsion, using a 1.9 cm (0.75 in) spreader nozzle attached to a 0.2 m\(^3\) min\(^{-1}\) (50 gal min\(^{-1}\)) pump. The full canal prism was sprayed with LA-PAM in both observation reaches, with only a partial prism application elsewhere. After application of LA-PAM, water was introduced to the canal on April 3, 2006. The partially hydrated form of LA-PAM had already been applied to
the Main Canal once in 2003 and to the Lower Mainline in 2004, and the emulsion form of LA-PAM had already been applied to the Upper Mainline once in 2005.

During the March 2006 LA-PAM application, the monitoring area consisted of Observation Reach 1 (8 km of the Main Canal), Observation Reach 2 (8 km of the Lower Mainline), and Buffer Reach (5 km of the Main Canal between the observation reaches). At normal capacity, discharge is typically 17 m$^3$ s$^{-1}$ (600 ft$^3$ s$^{-1}$) and 7 m$^3$ s$^{-1}$ (250 ft$^3$ s$^{-1}$) in Observation Reach 1 and Observation Reach 2, respectively. LA-PAM was not applied to Buffer Reach.

The sampling scheme was based on duplicate samples taken 61 m (200 ft) apart. Control samples were collected 30 and 90 m (100 and 300 ft) upstream and downstream of the LA-PAM treatment/non-treatment boundary in Observation Reaches 1 and 2. Within the application area, samples were collected 30 m (100 ft) upstream and 30 m (100 ft) downstream of the mid-point of Observation Reaches 1 and 2, and 30 and 90 m (100 and 300 ft) upstream of the lower LA-PAM treatment/non-treatment boundary. Samples were also collected at 30 and 90 m (100 and 300 ft) one-third and two-thirds of the way between Observation Reaches 1 and 2. At each site, three samples were collected: A) one 50-mL centrifuge tube for LA-PAM; B) one gallon jug for LA-PAM; and C) one 250-mL container for AMD. Two samples were collected in pairs, 61 m apart. These samples were collected based on the likelihood that high sediment concentration in the leading edge would remove LA-PAM from solution during the 24-hr hydration period for LA-PAM samples collected in the gallon jug. The gallon jug samples sat undisturbed overnight to permit full hydration of LA-PAM prior to subsampling and centrifuging. Linear anionic PAM was below detection limits at all sites except PAM-4, which was located 1.9 canal km (1.2 mi) within the second treated reach.

Figure 2-6. Map of the GVIC reaches where LA-PAM was applied in 2006.
2.4.3 Huntley Irrigation District, Billings, MT

The Huntley Project is located in south-central Montana and is managed by the Huntley Irrigation District (HID). Water was first delivered in 1908, but the project was extended to its current length in 1915. The canal diverts water south from the Yellowstone River to support agriculture between Huntley and Pompeys Pillar, MT. The main canal is 52 km (32 mi) long, has an operating capacity of 21 m$^3$ s$^{-1}$ (730 ft$^3$ s$^{-1}$), and irrigates approximately 121 km$^2$ (30,000 ac) through 322 km (200 mi) of laterals. The principal crops are alfalfa and other hay crops, sugar beets, silage, irrigated pasture, and small grains (U.S. Bureau of Reclamation, http://www.usbr.gov/dataweb/html/huntley.html).

Linear anionic PAM was applied to two adjacent reaches of HID (Figure 2-7A). At both sites, the polymer was added from a stationary location to observe downstream PAM-sediment floc interactions. Surface water measurements were conducted to assess seepage rate changes in response to LA-PAM application during the days prior to and after LA-PAM application. On the eastern reach (HID-1; Table 7-1), application began at Road 8 on July 14, 2006, and lasted for nearly two hours. Water samples were collected from three sites ranging from 0.8 to 7.4 km downstream. On the western reach (HID-2; Table 7-1), application began at Road 4 on July 16, 2006, beginning at 9:00 a.m. and lasting for two hours. On this reach, samples were collected at 1.9 and 3.6 km downstream.

The rate of LA-PAM application was equivalent to 1,222 and 1,099 kg km$^{-2}$ (10.9 and 9.8 lbs ac$^{-1}$) in the eastern (HID-1) and western reaches (HID-2), respectively. The application rate and time period of application were calculated with the assumption that LA-PAM applied at the upstream boundary would travel to the bottom of the treatment reach with the average water velocity. The 4.3 km (2.7 mi) long eastern treatment reach (HID-1) stretched from immediately downstream of lateral C to immediately upstream of lateral C. This reach had an average wetted perimeter of 16 m (52 ft) and water velocity of 2.1 ft s$^{-1}$. The 4.5 km (2.8 mi) long western treatment reach (HID-1) stretched from Road 4 to just upstream of lateral C. The western reach had an average wetted perimeter of 17 m (56 ft) and a water velocity of 0.7 m s$^{-1}$ (2.3 ft s$^{-1}$).
Kannah Creek Ditch #2, Grand Junction, CO

Kannah Creek Ditch #2 (KC) is a low-volume irrigation ditch located in Mesa County in western Colorado. Water is diverted from Kannah Creek, a small creek fed by snowmelt and springs at higher elevations from the Grand Mesa National Forest, terminating at the Gunnison River. Flows in KC are determined by stakeholder need, typically ranging between 3 and 10 cfs, up to a maximum of 15 cfs. Discharge was continuously measured using automated stage recorders placed in stilling wells associated with a pre-existing 3-ft Parschall flume at Site A and 2.5-ft H-flumes installed at Sites B and C for this study (Figure 2-7B). In 2005, LA-PAM was applied beginning at 10:30 am on July 18. Approximately 30 g of LA-PAM was applied every 100 feet to the canal starting at Site C and moving upstream to Site B. Water quality samples were taken at 30, 122, and 2816 m downstream of Site B. To increase the potential effectiveness of the polymer addition, a backhoe was used to dredge up sediment approximately 30 to 60 m upstream of point of current LA-PAM application. In 2006, LA-PAM was applied from Site C through Site A following similar methodology. Water quality samples were not collected during this second experiment.
Figure 2-7B. Kannah Creek Ditch #2 in western Colorado. LA-PAM was applied between sites B and C in 2005 and between sites A and C in 2006.

2.4.5 Lamar Canal, Lamar, CO

The Lamar Canal (LAM) diverts water from the Arkansas River just north of the town of Lamar in southeastern Colorado and runs approximately parallel to the south side of the river. An 11.9-km (7.4-mi) study reach, located south of Colorado Highway 50 between Lamar and Granada and extending from upstream site 400 to downstream site 405 (Figure 2-8) was selected for LA-PAM application (Figure 2-8). The LAM was different from the other two LARV canals studied in 2006 because it typically transports a much lower suspended sediment load and has lower turbidity. It is diverted from the Arkansas River just downstream of John Martin Reservoir, where much of the sediment from the river water settles.

Two LA-PAM application experiments were conducted at the LAM (LAM-1 and LAM-2; Table 7-1). The first application was conducted on the upper reach on May 18, 2006, and lasted for 8.3 hours. Water chemistry samples were only collected at a point just below the downstream boundary of the LA-PAM application zone (Figure 7-7). Granular
LA-PAM was added using hand spreaders. Suspended sediment was augmented by hand as four technicians raked the side of the canal approximately 10 m above the active point of LA-PAM application. Sediment addition by this technique increased suspended sediment from an estimated 67 mg L\(^{-1}\) to 2,900 mg L\(^{-1}\), as determined by several depth-width integrated samples taken throughout the experiment.

The second application was done on a lower reach on June 7, 2006, starting at 8:20 a.m. and lasting 9 hours. Though LA-PAM was applied to the entire study reach, irrigation diversion schedules caused all post-PAM measurements to be taken over a 9.3-km (5.8-mi) canal segment between sites 401 and 405. The application began at site 405 and a motorized boat and two hand spreaders were used over the majority of the reach. The boat motor stalled approximately 10.8 km (6.7 mi) upstream of site 405, so two technicians with hand spreaders walked the remaining 1.1 km (0.7 mi) to apply LA-PAM. A total of 91.1 kg (201 lbs) of LA-PAM were added to the canal water. The actual application rate was 12.6 kg ha\(^{-1}\) of LA-PAM (11.2 lbs ac\(^{-1}\)) or 7.7 kg km\(^{-1}\) (27.2 lbs mi\(^{-1}\)).

Figure 2-8. LA-PAM application study reach at the LAM in summer 2006.
2.4.6 Minnesota Ditch, Hotchkiss, CO

The Minnesota Ditch (MD) is in Gunnison County, Colorado, and diverts water for agricultural uses from the Minnesota Creek. The upper section of MD was chosen for LA-PAM application because it is perched on the hillside, has few outfalls, and has a general absence of agricultural return flows to the ditch that characterize lower sections of the ditch. Three flumes with automated dataloggers exist below the head gate in the control section, and at 3 and 5 km (2 and 3 mi) downstream of the study area. The upper flume provided discharge data in the control section, while the lower two flumes only provided ancillary discharge data due to the numerous unmeasured outfalls and return flows that impacted the ditch. Manual discharge measurements were taken at several locations and times during and after the LA-PAM application.

On June 23, 2005, LA-PAM was applied in granular form every 3 m (10 ft) with hand seed spreaders. Linear anionic PAM application began at the bottom of the 2.8 km (1.75 mi) treatment section and proceeded upstream to the bottom of the control section. Application began at 11:00 a.m. and continued for 2.5 hours. Discharge was initially monitored every 15 to 30 min (for the first three hours and then less often) at locations roughly 30 m, 150 m, 0.8 km, 1.6 km, and 2.8 km downstream of the LA-PAM application. Linear anionic PAM and SSC samples were also collected at each discharge measurement location every five to 15 minutes. Samples were collected and processed using the methods in section 3.4.

2.4.7 Rocky Ford Highline Canal, Rocky Ford, CO

The Rocky Ford Highline Canal (RFH) is an agricultural water delivery canal located near Fowler in southeastern Colorado (Figure 2-9). The entirety of its water comes from the Arkansas River watershed. Two LA-PAM applications were completed on the RFH in 2006.

The first application (RFH-1) took place on June 29 to 30, 2006, and stretched 32.0 km (19.9 mi) from upstream site 201 to downstream site 206 (Figure 2-9). The canal segment between sites 200 and 201 was used as an untreated control reach. This was the longest stretch of canal studied in 2006 and the application process required two days to complete. For the majority of this test, LA-PAM was applied from an automated spreader mounted on a boat. Operation failure of the boat occurred at 16.7 km (10.4 mi) upstream. An additional 2.6 km (1.6 mi) were completed by two technicians walking with hand spreaders. The first day of application took a total of nine hours and 25 minutes to complete. On the second day, the application was started back at km 16.7 (mi 10.4), where the boat had malfunctioned on the previous day. The second day of application took approximately four hours and 35 minutes and covered 16.4 km (10.2 mi) of canal. A total of 408 kg (901 lbs) of LA-PAM was applied over the two days at a rate of 13.3 kg ha\(^{-1}\) (11.9 lbs ac\(^{-1}\)) or 12.8 kg km\(^{-1}\) (45.3 lbs mi\(^{-1}\)). Water samples were collected 0.5 km and 2.1 km downstream of the lower polymer treatment reach boundary on the first day.
A second application at the upper reaches of this canal was conducted on July 20, 2006, between sites 201 and 202. This reach consistently displayed greater seepage rates relative to the downstream reaches previously studied on the RFH. In addition, LA-PAM was applied 2.4 km (1.5 mi) upstream of site 201 to ensure adequate hydration of LA-PAM and sediment flocculation prior to reaching site 201. A total length of 6.1 km (3.8 mi) was treated with LA-PAM. This application followed a storm event that created high SSC in the canal. Turbidity was 270 NTU at the start of application process, and was slowly returning to baseline levels. As in the first application, a boat with an automatic spreader was used. Completion took 50 minutes, as there were few obstructions in the canal segment and increased boat speed. A total of 111 kg (245 lbs) of LA-PAM was applied at a rate of 14.2 kg ha⁻¹ (12.7 lbs ac⁻¹) or 18.2 kg km⁻¹ (64.4 lbs mi⁻¹) during two application periods occurring on subsequent days.

A third LA-PAM application on the RFH was scheduled for August 2007, but was cancelled due to low seepage rates caused by natural sediment deposition following an unusual storm that delivered a significant load of fine suspended sediment into the canal. This natural reduction in seepage rates lasted several weeks. When seepage rates returned to near pre-event levels, turbidity in the canal had dropped below the minimum threshold (80 to 100 NTU) thought to be needed for an effective treatment. Turbidity remained at similar levels throughout the remainder of the water year.
Although conditions did not permit a moving LA-PAM application in 2007, a stationary LA-PAM application was performed at Road 5.25 on August 16, 2007 (RFH-4). Twenty-three kg (50 lbs) of LA-PAM were applied at a rate of 0.25 kg min⁻¹ during a 91-minute period using a hand fertilizer spreader located at the center of a bridge. At the point of application, the canal had an estimated wetted perimeter of 9 m (30 ft) and a discharge of 4.2 m³ s⁻¹ (150 ft³ s⁻¹). Grab samples for turbidity and SSC were collected from eight sites up to 4.5 km downstream of the application point (Table 2-3). An additional site 6.1 km downstream contained a continuous turbidity meter (DTS-12, FTS, Inc., Victoria, BC, Canada). Rather than a strict time-based approach, sampling was concentrated to times when visually observable changes in turbidity were noted. Control samples were taken from the center of the upstream side of the LA-PAM application site. Samples at the control and sites 1, 5, 6, 7, and 8 were collected with a 0.95-L, depth-integrated, isokinetic sampler. An open-mouthed, plastic 1-L jar was used to manually collect depth-integrated samples from the thalweg at sites 2, 3, and 4. Immediately after collection, subsamples were taken for turbidity and LA-PAM concentration. Turbidity was measured using a LaMotte 2020e within 36 hours. Linear anionic PAM samples were centrifuged within 36 hours following standard procedures (Section 3.4).

Table 2-3. Sampling locations during the stationary application experiment at the RFH, CO. A continuous turbidity sensor was installed in the canal at the furthest site (“Auto TU”). Datum: WGS 84.

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<thead>
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<th>Location</th>
<th>Latitude</th>
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</tr>
</thead>
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</table>

2.4.8 Smith Ditch #7, Grand Junction, CO

Smith Ditch #7 (SD) is located in Mesa County, Colorado, approximately 32 km (20 mi) south of Grand Junction. The ditch originates at a diversion from Kannah Creek (in the lower Gunnison River basin) and flows approximately 8 km (5 mi) before becoming fully used on irrigated land. Water flows only during the irrigation season (May to October) and is not used for human consumption. Because it dries annually, SD does not support fish populations and is poor-quality aquatic habitat. Maximum discharge of water within the ditch was 0.2 m³ s⁻¹ (8 ft³ s⁻¹) during early season and was approximately 0.1 m³ s⁻¹ (5 ft³ s⁻¹) during the experiment.
Linear anionic PAM was applied to SD on August 25, 2005, beginning at 10:00 a.m. for a duration of just over three hours. Approximately 11 kg ha\(^{-1}\) (10 lbs ac\(^{-1}\)) of LA-PAM were applied to 1.6 km (1.0 mi) of SD on August 25, 2005, with an equally long control reach upstream of the application (Figure 2-10). A backhoe was used to stir the canal bottom, temporarily increasing suspended sediment loads. Three 0.45-m (1.5-ft) H flumes were installed to monitor stage at 10-minute intervals during the application. Stage within the flumes was continuously monitored with either a Global Water datalogging pressure transducer (Gold River, CA) or a KPSI transducer (Hampton, VA) that was datalogged by a datalogger (model CR205, Campbell Scientific, Inc., Logan, UT).

Aquatic benthic macroinvertebrates (BMI) in the substrate were sampled before and after PAM application, and drift macroinvertebrate communities were sampled before, during, and after LA-PAM application according to the methods described in Section 3.3. These measurements were taken on a daily basis for four days before and for five days after LA-PAM application.

Water samples for LA-PAM and SSC were collected at eight sites: 608 m into each of the control and treatment reaches, and 0, 100, 250, 500, 940, and 1,379 m downstream of the LA-PAM application. The frequency of sampling was determined by the turnover time; that is, the time it takes for the last slug of water to which LA-PAM is applied to travel past the last sampling point downstream. Samples for LA-PAM analysis were collected on a 15-minute basis within the first turnover and on a 30-minute basis within the second turnover at the 0- and 100-m sites to better define the boundary conditions required for the models of LA-PAM fate and transport. All other sites (250, 500, 940, and 1,379 m) were sampled every 30 minutes within the first turnover and on a 60-minute basis within the second turnover. Suspended sediment concentration samples were collected every hour at each of the sites. Additional LA-PAM and SSC samples were collected twice during the day following application, and once per day for six subsequent days at most sites. Samples for AMD analysis were collected only at the 0-m site just below the lower end of the PAM treatment area.
reach. All water quality samples were collected according to the methods described in Section 3.4.

Discharge and surface slope measurements were conducted at three sites (0, 500, and 940 m downstream of the treatment area) before and after the application of LA-PAM using a hand-held flowmeter with wading rod or total station. Bed material samples were collected at these three locations and three more locations in the treated reach before LA-PAM was applied. Samples were collected using a shovel since the water depth was shallow.

2.4.9 South Canal, Uncompahgre Irrigation District, Delta, CO

The largest canal to which LA-PAM was applied was the South Canal of the Uncompaghre Project (UNC; Table 7-1) located in west-central Colorado. The South Canal carries water diverted from the Gunnison River to the Uncompahgre River. At the time of the experiment, the South Canal delivered an additional 790 cfs into the Uncompahgre River that was carrying only 350 cfs, measured 5.6 km upstream (USGS Station 09147500). The main objective of this experiment was to assess the response of benthic macroinvertebrates in the Uncompahgre River to the exposure of canal water treated with LA-PAM. Application of LA-PAM began on August 15, 2006, and lasted for 2.3 hours. A motorboat and automated spreader were used to apply LA-PAM to a 6.3-km reach of the UNC. Water chemistry and BMI samples were collected from the South Canal downstream of the polymer application zone and in the Uncompahgre River both above and below the outlet of the South Canal.
3.0 FIELD AND LABORATORY METHODS

3.1 Seepage Estimation

In 2005 and 2006, seepage estimates were conducted at least the day before and the day after LA-PAM application. Seepage was estimated by comparing surface water discharge above and below the reach of interest and accounting for all other inflows and outflows along the reach. For the smaller canals (<0.4 m³ s⁻¹ [15 ft³ s⁻¹]), 0.5- or 0.6-m (1.5- or 2-ft) H flumes with pressure transducers were used to estimate discharge. Auxiliary hand measurements were taken by either a Marsh-McBirney Flo-Mate (Frederick, MD) or a Global Water FP201 (Gold River, CA) velocity meter using standard methodology (Laenen 1985; http://pubs.usgs.gov/twri/twri3-a17/). Seepage rate estimates were focused on short-term reduction at these sites and were based on the total volume of water entering and exiting the treatment reach through flumes, accounting for the travel time of canal water. For the larger canals, surface water discharge was measured with a Teledyne-RDI StreamPro (2007) (Poway, CA) acoustic Doppler current profiler (ADCP) with auxiliary measurements taken using a Sontek FlowTracker (San Diego, CA) acoustic Doppler velocimeter (ADV). The methods discussed below were followed for both short- and long-term measurements conducted in LARV canals in 2006 and 2007.

3.1.1 Volume Balance Procedure for Estimating Seepage

The effectiveness of an LA-PAM application was evaluated by calculating and comparing the seepage rates pre- and post-treatment. Inflow, outflow, and storage change rates were measured or calculated and used to estimate the total seepage rate, \(Q_s\) (m³ s⁻¹), through the wetted perimeter of the canal reach over a time period \(\Delta t\) as:

\[
Q_s = Q_{US} - Q_{DS} + Q_I - Q_D - Q_E - \frac{\Delta S}{\Delta t}
\]

where \(Q_{US}\) is the canal inflow rate through the upstream cross section (m³ s⁻¹), \(Q_{DS}\) is the canal outflow rate through the downstream cross section (m³ s⁻¹), \(Q_I\) is the total rate of inflows along the canal reach (m³ s⁻¹), \(Q_D\) is the total rate of outflow diverted along the reach (m³ s⁻¹), \(Q_E\) is the total rate of evaporation from the water surface along the reach (m³ s⁻¹), and \(\frac{\Delta S}{\Delta t}\) is the rate of change of stored water volume within the canal reach (m³ s⁻¹).

Equation (1) was used to estimate seepage losses for all measurements before and after LA-PAM application in each canal study reach. For example, at LAM on June 6, 2006, the estimated value of \(Q_{US}, Q_{DS}, Q_I, Q_D, Q_E\), and \(\frac{\Delta S}{\Delta t}\) was measured as 1.42 m³ s⁻¹ (50.0 ft³ s⁻¹), 1.01 m³ s⁻¹ (35.8 ft³ s⁻¹), 0.0 m³ s⁻¹ (0.0 ft³ s⁻¹), 0.10 m³ s⁻¹ (3.6 ft³ s⁻¹), 0.02 m³ s⁻¹ (0.7 ft³ s⁻¹), and 0.01 m³ s⁻¹ (0.4 ft³ s⁻¹), respectively. Upon applying these volumetric rates to Equation (1), a seepage rate of 0.28 m³ s⁻¹ (9.7 ft³ s⁻¹) was calculated. The total wetted perimeter area of the study reach was estimated at 5.7 ha (14.0 ac), making the estimated seepage rate 0.05 m³ s⁻¹ ha⁻¹ (0.5 ft³ s⁻¹ ac⁻¹). The methods for estimating each of the individual variables are described in the following sections.
3.1.2 Canal Flow Rate Measurements, $Q_{US}$ and $Q_{DS}$

Values of $Q_{US}$ and $Q_{DS}$ were measured for each study reach using a StreamPro ADCP or a FlowTracker ADV. Both types of equipment use an area-velocity technique to estimate the flow rate, in which the cross-section averaged flow velocity perpendicular to the cross section is multiplied by the cross-sectional area of flow to obtain a volumetric flow rate.

Disadvantages of the ADV are lengthy measurements and a limited number of velocity measurement points in a cross section. Compared to the ADCP, the ADV is not nearly as time efficient, which limits the quantity of data that can be collected in a given time period, $\Delta t$. Storage changes can potentially occur during the lengthy ADV measurements, making data analysis more complicated and prone to error. However, comparisons of these two flow measurement technologies found that they give similar results under stable flow conditions. In 2006, only one ADCP was available for use, so both technologies were used for discharge measurements. Using both the ADV and the ADCP allowed for increased data collection and a comparison of results between the two technologies. In 2007, three to four ADCPs were available and all flow rate measurements were conducted with an ADCP.

3.1.2.1. ADV Discharge Measurement Technique

Rigid pins (rebar or T-posts) were placed perpendicular to water flow on the left and right canal banks and a tape measure was fastened to the pin on each bank. In an attempt to improve flow measurement accuracy, a three-point measurement technique (i.e., heights of 0.2, 0.4, and 0.8 times the total flow depth) was used for all flow depths greater than 0.30 m (1 ft). Depth-averaged flow velocity was then calculated by the ADV at each station using a weighted average equation where the flow velocity at 0.4 of the total flow depth was weighted twice as much as the flow velocities at 0.2 and 0.8 of the total depth. A specific conductance probe was used to adjust for signal travel velocity variations due to water salinity. The flow area at each station was calculated by the FlowTracker, based on the distance from the starting location of the measurement and total flow depth. Multiple FlowTrackers were available for use in 2006, so measurements were completed simultaneously at each cross section within a study reach. In this way, $Q_{US}$ and $Q_{DS}$ for the study reach control volume were measured approximately over the same time period.

Uncertainties in ADV measurements can arise due to turbulent flows, low scatter (sediment) in the water, interference of the acoustic signal caused by floating debris or channel boundaries, inaccurate estimations of electric conductance affected by salinity concentrations, skewed cross sections not perpendicular to the channel banks, velocity spikes, and low signal-to-noise ratios (SonTek/YSI, 2003). FlowTracker ADVs are limited in the amount of data they collect when selecting the operational mode that automatically calculates discharge. Velocity profiles are developed based upon a maximum of three velocity readings within a water column. For turbulent flows or inverted velocity profiles, three velocity readings may not accurately represent average water column velocity. This causes uncertainty in the velocity-area method used by ADVs to calculate total flow rate over the cross section. The three-dimensional acoustic sensor is mounted onto a metal wading rod that is held stationary on the channel ground for velocity measurements. Specifically for soft channel beds such as sand, the wading rod can sink below the ground surface or shift from forces created by flowing water. This can affect the estimated total depth of the water column.
that is entered into the ADV software and consequently affect the depths at which velocity readings are collected. Despite these uncertainties, the ADV FlowTracker on average has been found to produce discharge measurements within 5 percent of the traditional mechanical propeller current meter technology when using the USGS methodology (Rehmel, 2007). The manufacturer of the FlowTracker suggests that individual velocity readings of the ADV are accurate within 1 percent of the true velocity (SonTek/YSI 2003).

3.1.2.2. ADCP Discharge Measurement Technique

The ADCP StreamPro consists of a data collecting transducer mounted on a flotation device that can traverse the water while remaining aligned with the principal direction of flow. At each cross section, a pulley system was set up perpendicular to water flow direction on both banks of the canal and the ADCP unit was allowed to travel freely as it was pulled at an even tempo across the channel (Figure 3-1). Data were transmitted wirelessly from the ADCP to a pocket PC monitored by a technician. The StreamPro collected point velocity and water depth measurements, and measured the distance each second as it traveled between banks. Data were collected for a total of at least 12 minutes (sum of the transect measurement times) as the StreamPro traversed multiple times from one bank to the other at an approximate steady speed. Due to signal instability and interference, the ADCP cannot collect velocity data close to the bottom of the channel bed, near the water surface, or near the channel banks, so these measurements were estimated based on input parameters such as bank geometry and distance to the bank. All measurements from one location were summarized into a single discharge measurement using the WinRiver software provided by the manufacturer.

Figure 3-1. ADCP StreamPro measurement at LAM.
Measurements were conducted simultaneously at each canal location or from upstream to downstream locations in succession. For shorter canal reaches, successive measurements were synoptic, meaning roughly the same parcel of water could be measured at the upstream and downstream boundaries of the reach. To ensure that storage changes within the studied control volume did not affect the accuracy of the results, water levels in the canal were monitored using pressure transducers or by the recorded flow rates at the head gates of the canal on the Colorado Department of Water Resources (CDWR) website (http://www.dwr.state.co.us) during the time of measurement. Measurements were rescheduled or repeated if unfavorable storage and/or unstable flow conditions existed.

The ADCP uses a Doppler approach to track the canal bed under the assumption that the bed is not moving; therefore, significant bedload motion can potentially affect measurement accuracy. To account for the associated error, a moving bed test was completed at each cross section by placing the unit at a stationary location near the center of the channel and collecting data for approximately 5 to 10 minutes. If the moving bed was not a significant source of error at a given cross section under certain measured flow rates, the moving bed test was no longer conducted at that cross section for similar flow rates in the future.

Uncertainties in ADCP measurements arise from several sources, including instrument limits (i.e., Doppler noise, spatial resolution, and side-lobe interference), user error (i.e., incorrect edge coefficient estimation or instrument operation), and the measuring environment (i.e., difficulty with bottom tracking or instrument rotation due to high turbidity or turbulent water) (Gonzalez-Castro and Muste, 2007). One benefit of ADCP technology is its ability to quickly collect a large quantity of data, although measurements may encompass temporal variation in the flow rate if transects are collected over a long time period.

3.1.3 Diverted Outflows, $Q_D$

Irrigation diversion structures were present in each of the canal study reaches. Canal reaches with few or no diversions were made a priority in the selection of study reaches and the time of measurement, although this proved difficult in light of the need for reaches to be long enough to have seepage rates greater than the expected cumulative error of the flow measurement methods. When possible, the diversions were shut off by canal managers, but occasionally diversions were active, depending upon the irrigation demands. Diversion rates were measured using a variety of techniques, including Parshall flumes within the diversion channels just downstream of the offtake, ADV FlowTracker measurements within the diversion channels, and/or diversion pump rating curves.

To ensure accuracy, both Parshall flume and ADV measurements were usually collected in the diversion channels. The measurements in Parshall flumes were determined by measuring the flow depth at the location of a staff gage within the flume and using a calibrated discharge rating curve to calculate flow rate. The result was then checked with an ADV measurement. Measurements using only an ADV FlowTracker were completed if no Parshall flume was present in the diversion channel. The open-channel flows in the diversion channels were often turbulent, affecting the accuracy of the ADV measurements, while the accuracy of flume measurements were affected by downstream submergence and by lateral or vertical settling (Abt et al., 1995). Despite these difficulties, results of the flume and ADV measurement were always very comparable, increasing confidence in the calculations and measurement methods.
In one case, the diversion was a pump and closed conduit (no open channel flow), and outflow was estimated by the canal manager based on a rating curve. Uncertainty in estimating pumped flow rates using rating curves can be substantial, especially as pumps age and wear down. Therefore, outflow estimations provided by the canal manager were checked on several occasions by placing a bucket of known volume under its release pipe and timing how long it took to fill the bucket. The discharge numbers provided by the canal manager and the measured flow rate using the buckets were very comparable. Diversion rates from the pump never exceeded 0.04 m$^3$ s$^{-1}$ (1.5 ft$^3$ s$^{-1}$) or 1 percent of $Q_{US}$; thus, errors minimally affected the seepage calculations.

The LAM study reach contained the most active diversions. Large diversions were bracketed in the canal by flow measurement cross sections in an attempt to accurately determine diversion discharges. Measured values of $Q_D$ ranged from 0.10 to 0.34 m$^3$ s$^{-1}$ (3.6 to 12 ft$^3$ s$^{-1}$) at LAM, or approximately 7 to 26 percent of the discharge at the upstream study boundary. These values were much greater than those typically observed at RFH or CAT. Diversions were typically shut off by RFH managers when seepage measurements were taking place. The maximum $Q_D$ ever active on the RFH during seepage measurements was 0.02 m$^3$ s$^{-1}$ (0.6 ft$^3$ s$^{-1}$), less than 1 percent of $Q_{US}$. One diversion was typically active at the CAT study reach. A second diversion was only active twice in 2006 and 2007. The total diversion rates at the CAT study reach ranged between 0 and 0.04 m$^3$ s$^{-1}$ (0 to 1.5 ft$^3$ s$^{-1}$), never exceeding 1.1 percent of $Q_{US}$.

3.1.4 Inflows, $Q_I$

Inflows from sources adjacent to irrigation canals in the LARV are not uncommon, especially drainage from land on the upper contours. Prior to site selection, locations of possible $Q_I$s were observed with the assistance of canal managers familiar with the region. Any $Q_I$ locations present within the study reach were checked on a regular basis for possible discharge into the canals during seepage measurements. No $Q_I$ was observed during the seepage studies conducted in 2006 and 2007.

3.1.5 Free-water Evaporation, $Q_E$

The Penman combination equation was used to estimate the depth rate of free-water evaporation, $E$ (m day$^{-1}$), from the LARV canals during inflow-outflow measurements:

$$E = \frac{\Delta(K + L) + \gamma K_E \rho_w \lambda_v v_e^* (1-W_s)}{\varphi_w \lambda_v (\Delta + \gamma)}$$  \hspace{1cm} (2)

where $E =$ Free-water evaporation rate (m day$^{-1}$);

$\Delta = $ Slope of relation between saturation vapor pressure and temperature (kPa K$^{-1}$)

$K =$ Solar radiation [MJ (m$^2$ day$^{-1}$)$^{-1}$]

$L =$ Longwave radiation [MJ (m$^2$ day$^{-1}$)$^{-1}$]

$\gamma =$ Psychrometric constant (kPa K$^{-1}$)

$K_E =$ Coefficient of vertical transport of water (kPa$^{-1}$)
\[ \rho_w = \text{Mass density of water (kg m}^{-3}) \]
\[ \lambda_v = \text{Latent heat of vapor (MJ kg}^{-1}) \]
\[ v_a = \text{Wind speed (m day}^{-1}) \]
\[ e_a^* = \text{Saturated vapor pressure (kPa)} \]
\[ W_a = \text{Relative humidity (expressed as a fraction)} \]

Data for estimating the parameters of Equation (2) were obtained from the Colorado State University (CSU) Colorado Agricultural Meteorological Network (CoAgMet) website, which lists hourly atmospheric data at numerous weather stations across Colorado. The data used for each of the three study canals were obtained from different weather stations, each within a few kilometers of the study reaches. Dingman (2002) presents equations and detailed definitions for the variables within Equation (2). The volumetric evaporation outflow rate, \( Q_E \), from a canal study reach was computed by multiplying \( E \) by the estimated water surface area of the study reach (the product of average top width and canal reach length).

Uncertainty in the evaporation rate may arise from uncertainty in the available data. Hourly data pose some error in evaporation calculations since they are not completely representative of the time periods in which seepage measurements were conducted. Atmospheric data collection technologies at the weather stations are subject to measurement error, and improper representation of the study reach conditions due to the distance to the closest weather station affect calculation accuracy. However, values of \( Q_E \) at CAT, LAM, and RFH canals were typically very low and ranged between 0.001 and 0.003 m\(^3\) s\(^{-1}\) km\(^{-1}\) (0.05 to 0.15 ft\(^3\) s\(^{-1}\) mi\(^{-1}\)). Values never exceeded 2 percent of \( Q_{US} \) and were typically significantly less than 1 percent of \( Q_{US} \). The ratio \( Q_E/Q_{US} \) (in percent) at CAT and RFH canal reaches was never greater than 1 percent and was typically less than 0.5 percent.

3.1.6 Storage Changes, \( \Delta S \)

In canal systems, flow rates and water levels are unsteady. Such temporal variability is due primarily to changes of flow rates and water levels in the river, changes of the settings of canal regulating structures, changes in diversion rates from the canal, changes in hydraulic geometry within the study reach, and atmospheric variability. Storage changes were estimated through two methods to monitor changing water levels during flow rate measurements. The first was through the installation of pressure transducers at selected locations within the canal. Using measured pressure and appropriate mathematical conversions, the height of the water column above the device was determined. The second method was installation of staff gages and manual reading of the water level from the gages at specific increments of time (typically every 5 to 15 minutes).

Sources of uncertainty with estimating storage changes are primarily associated with errors in stage readings and measurements of channel geometry. For pressure transducers, uncertainty exists in the measurement as well as in the conversion from absolute pressure to gage pressure based upon a separately measured atmospheric pressure. Manual stage readings from staff gages are affected by wave action, flow velocity, and an observer’s ability to interpret the water level. Other sources of uncertainty in seepage measurements include: estimating the locations over which the storage change occurs in the control volume,
the time duration of the change, and the total change in storage volume. Attempts were made to limit the errors associated with these uncertainties by collecting frequent stage data, characterizing the channel geometry with closely spaced cross sections, and, most importantly, conducting seepage measurements during near-steady-state flow conditions.

By monitoring flow rates from the CDWR (http://www.dwr.state.co.us), monitoring stage data from staff gages and pressure transducers, and communicating with canal managers, significant storage changes during flow measurements could be avoided in most cases. Storage changes on the RFH were minimal since the study reach was located immediately downstream of the canal headgate, which regulates fairly steady flow rates entering the canal from the Arkansas River. The CAT and LAM study reaches were located downstream of numerous irrigation diversions; thus, the water level in the canals had to be monitored carefully to identify and quantify storage changes.

3.1.7 Determining Hydraulic Geometry Characteristics

Starting in spring 2007, survey equipment (e.g., a level and total station) was used to collect canal geometry data. Enough geometry data were collected to provide preliminary hydraulic geometry characteristics of the CAT and RFH canals. More surveying is being done to provide more accurate estimations of canal geometry. In addition to cross-section surveys with survey equipment, ADV FlowTracker and ADCP StreamPro measurements provide cross-sectional bed profile data. These data also can be used to determine hydraulic geometry values at every flow measurement location. Cross-sectional surveys every 0.16 km (0.1 mi) within a study reach are currently being completed using surveying equipment to reduce the importance of individual cross sections and errors associated with hydraulic geometry.

At each surveyed cross section, wetted perimeter, cross-sectional flow area, and the top width of the free water surface were estimated for the time periods in which seepage measurements were conducted. Wetted perimeter values are very important in estimating seepage reduction because all reduction percentages are reported in reference to seepage rate per unit wetted perimeter area of the canal (i.e., m$^3$ s$^{-1}$ ha$^{-1}$ of wetted canal) to account for variance in seepage due to flow rates and canal geometry. A weighted average mathematical technique was used to calculate total wetted perimeter area for a study reach based upon the wetted perimeter at each surveyed cross section. Each cross section was assumed to represent the canal prism extending between that cross section to the midpoint of the adjacent cross sections (Figure 3-2). With known values of wetted perimeter and channel length between cross sections, a total wetted perimeter area of the study reach, $Area_{wp}$, was computed as

$$Area_{wp} = P_i \left( \frac{L_{i,i+1}}{2} \right) + P_{i+1} \left( \frac{L_{i,j+1} + L_{i+1,j+2}}{2} \right) + P_{i+2} \left( \frac{L_{i+1,j+i+2} + L_{i+2,j+i+3}}{2} \right)$$

$$+ P_{i+3} \left( \frac{L_{i+2,j+i+3} + L_{i+3,n}}{2} \right) + \ldots + P_n \left( \frac{L_{i+3,n}}{2} \right)$$

(3)

where $L_i$ and $P_i$ represent channel length between cross sections, $i$, and wetted perimeter at a specified cross section, respectively.
Figure 3-2. Depiction of a canal study reach.

Top width, $T_w$, was used to calculate the evaporation rates from the canal free water surface over each seepage measurement time period. The total water surface area, $A_{area}$, in a study reach was computed as

$$
A_{area} = T_w \left( \frac{L_{i,i+1}}{2} + T_{w_{i+1}} \left( \frac{L_{i+1,i+2} + L_{i+2,i+3}}{2} \right) + T_{w_{i+2}} \left( \frac{L_{i+2,i+3} + L_{i+3,n}}{2} \right) \right) + T_{w_{i+3}} \left( \frac{L_{i+2,i+3} + L_{i+3,n}}{2} \right) + T_{w_{n}} \left( \frac{L_{i+3,n}}{2} \right)
$$

An interpolation process was completed to estimate the stage at each surveyed cross section based upon the longitudinal profile and the stage data collected by each pressure transducer. To account for the lack of pressure transducer data at some sites, a longitudinal survey of each canal was conducted to relate each of the cross-sections elevations and stage data.

3.1.8 Temperature as a Tracer for Determining Seepage

In addition to surface water measurements, seepage rates were investigated at exact locations (as opposed to full canal reaches) using temperature as a tracer for surface-ground water exchange. Dowel rods instrumented with thermocouple probes were installed in selected locations. Thermocouple probes were made up of two dissimilar metal wires, joined together at one end. When the junction between the two wires was heated or cooled, a voltage was produced that translated by a datalogger into a temperature. Patterns in the temperature measurements at depth were correlated with the amount and velocity of water traveling across the thermocouple, so seepage was determined through analysis of thermal amplitude fluctuation with depth, as well as the lag time between periods at different depths. This approach has been used in natural streams and rivers (Constantz et al., 2002; Stonestrom and Constantz 2003; Hatch et al., 2006) and was found to produce seepage estimates that were consistent with ponding studies conducted in the Truckee Canal in Fernley, NV (Mihevc et al., 2002).

In May 2006, dowel rods instrumented with thermocouple probes were installed at four locations spread out along 20 miles in the RFH. The sites were selected based on visual inspection (areas where downgradient vegetation was particularly lush or included species known to prefer wet areas suggesting local seepage) and discharge estimates. Each of these locations underwent up to two LA-PAM applications during summer 2006.
At each site, thermocouple probes were attached to the dowel rods at five depths, with the top probe approximately at the sediment-water interface and the lowest probe 1.1 m (3.5 ft) into the sediment (Figure 3-3). The probes were attached to dataloggers (models CR10X or CR1000, Campbell Scientific, Inc.). An additional dowel was installed at an angle into the canal bank to determine differences between vertical and lateral seepage rates. Water and soil temperatures were included for reference. Temperatures have been logged continuously at 20-minute intervals at these four locations since installation.

Figure 3-3. Position of thermocouples in the canal bed.

In August 2007, an additional set of eight dowel rods were installed at six locations longitudinally within a 30-m (100-ft) length of the Limestone Division of the FL, where high seepage was apparent (there is an adjacent wetland on the downgradient side). Data from these probes will show the extent of local heterogeneity within a short reach of canal bed and predict the ability to upscale seepage estimates at certain locations to longer reaches of the canal.

Using temperature as a tracer has several advantages over other traditional methods, such as surface water discharge measurements. Most importantly, this approach produces a continuous estimate of changes in vertical water flow over time, providing the ability to estimate seepage when field personnel are not available, or when environmental conditions are not suitable for surface water measurements (e.g., high flows, impassable canal access roads). In addition, the ability to estimate seepage reduction using the tracer method is not affected by the factors (e.g., high turbidity, unfavorable canal geometries and bed composition) that prevented acoustic Doppler-based (e.g., Teledyne-RDI StreamPro, 2007) surface water discharge estimates from being taken for significant portions of the 2007
agricultural season. As the thermal tracer approach is based on automated, near-continuous data collections, it should be suitable to assess the changes in seepage due to point events, such as the application of LA-PAM, or due to “self-sealing” in response to high suspended sediment loads mobilized by thunderstorms. Additionally, this method can indicate how seepage naturally changes throughout the irrigation season, but more research is necessary to understand the time-scale and link smaller seasonal changes in bed hydraulic properties to seepage estimates.

3.2 AMD Transport into Groundwater: Dye Tracer Study

Existing wells along the CAT and RFH canals were evaluated for a tracer test study. Criteria used for site evaluation include proximity of wells to the canal, the ability of a well to produce water, and the inferred hydraulic conductivity of the sediments near these wells. Site evaluation began with well development of existing wells. Monitoring wells were developed with a bailer and a small-diameter submersible pump. Several well volumes were removed from the monitoring wells and recovery times were noted. During this process, many of the well transects were determined to be unsuitable for the tracer test due to long recovery times and excessive distance between the wells and the canal. The only suitable site for the tracer study was the CAT 5 transect. Although the three wells present had desirable hydraulic characteristics, the well spacing was still too great to perform the tracer test. To use this site two additional wells were installed (see Figure 3-4).

Figure 3-4. Cross section of the CAT Site 5. Water levels shown were measured before the tracer test on August 7, 2007.
Monitoring wells were drilled along the CAT and RFH canals in spring 2007 to support hydraulic investigations for canal seepage studies. These monitoring wells were drilled with a Giddings tractor mounted auger rig. Boreholes were 12.7 cm (5 in) in diameter and drilled to depths between 2.4 and 6.0 m (7.7 and 9.7 ft). Wells were completed with 7.6-cm (3-in) casing that was slotted from the bottom of the borehole to just below land surface. Slotted screen was backfilled with sand where borehole conditions allowed. Surface completion consists of blank, nonslotted casing from just below land surface to slightly less than a meter above land surface. Blank casing was backfilled with bentonite to preclude infiltration of surface water. Wells were drilled in transects perpendicular to the canal. Transects consist of three to five wells ranging from 4.5 to 46 m (15 to 150 ft) from the canal. Generally, the well separation was too large to be useful for a short-duration tracer test.

Well development at Site 5 indicated the hydraulic properties of the formation were suitable for a tracer test. However, because of the large spacing between the wells and the canal, two additional wells were installed (Figure 3-4). This first well installed, 5D, was placed between the canal road and the canal. When this well was drilled to a depth of 2.4 m, no water was encountered. The well was then deepened to 3.5 m, where sufficient water was encountered to make this a good monitoring well. Because of the close proximity of well 5D to the canal (2 m), it was not possible to install a well between 5D and the canal to serve as an injection well. To allow for tracer injection, a well was installed in the canal. A hand auger was used to install the injection well to a depth of approximately 2 m (6.6 ft) below the canal. Well casing was constructed with 0.3 m (1 ft) of slotted pipe and the remainder blank pipe. Bentonite was packed between the blank pipe and the borehole to isolate the injection well from the canal. An effective seal was noted as the water level in the injection well was 1.05 m below the water level in the canal. A level-line survey was conducted at this site to determine water elevations and hydraulic gradient. At this site, the hydraulic gradient is 0.04 m m⁻¹.

Because it was desirable to conduct two simultaneous tracer tests and the available well sites were not suitable, a new transect of wells was installed for the second tracer test. The second site was chosen based on observed canal leakage. Several sites were identified as leaking sites by vegetation and swampy areas downgradient of the canal. After several sites were identified, Site 2 was chosen for the tracer test because of the observed leakage and ample space to install several wells. Five wells were installed at Site 2. Sediments encountered in these wells consist of silts and sands with some thin clay layers. Wells were drilled to approximately 3 m (10 ft), with water levels ranging from 0.45 to 1.4 m (1.5 to 4.5 ft) below land surface. The monitoring wells were drilled and completed with the same technique as the other monitoring wells in the study area. The monitoring wells were developed with a bailer followed by a small submersible pump. All of these wells exhibited high conductivity, evident by the fact that they could not be pumped dry during development (Figure 3-5). Wells were named with letters by site, with well 2A being adjacent to the canal and well 2E located the furthest from the canal. The height of each well was determined with a level-line survey. From this survey, relative well heights and water elevations were determined using the canal road as datum. The hydraulic gradient observed in these wells is relatively steep at 0.10 m m⁻¹.
Rhodamine WT liquid (Keystone Aniline Corporation, Chicago, IL), a dye tracer commonly used in groundwater tracer tests, was chosen as the tracer in this investigation. Rhodamine concentrations can be determined quantitatively using a fluorometer and then qualitatively by visual inspection. To accelerate the movement of groundwater during the tracer test, submersible pumps were installed in wells 2D and 5D. Discharges from these pumps were 3.8 L\(^{-1}\), directed away from the monitoring wells so they would not interfere with the test. These pumps were also used to collect the samples for wells 2D and 5D.

![Cross section of the CAT Site 2](image)

Figure 3-5. Cross section of the CAT Site 2. Water levels shown were measured before the tracer test on August 7, 2007.

On August 6, 2007, a preliminary tracer test utilizing fluorescein dye was conducted at Site 5 to determine the travel time from the injection well to well 5D. Fluorescein is a yellow dye tracer that was used qualitatively (visually) for this test because it can easily be distinguished from the red dye used in these wells during the LA-PAM treatment to the adjacent canal. The dye was injected at 13:05 hrs in the injection well and a submersible pump was started in well 5D. During the first day, the tracer was not observed in well 5D. However, on August 7, 2007, during the tracer test using the rhodamine dye, fluorescein appeared 60 minutes into that test, which was 22 hours after the injection. The concentration continued to increase over the duration of the tracer test on August 7. The following day, fluorescein was not observed in the samples from well 5D.

The tracer test was conducted on August 7, 2007, in conjunction with the application of LA-PAM at CAT. The section of the canal where LA-PAM was applied began at site 202 and ended at site 201 (Figure 2). Linear anionic PAM was applied from a small boat with two hand spreaders. When the boat passed Site 5 at 10:21 hrs, approximately 2 L of liquid tracer were poured into the injection well and the pump in well 5D was started. While the boat was in transit between Site 5 and Site 2, it broke down and was unable to proceed for about an hour. Once repaired, LA-PAM application continued and the boat passed Site 2 at 12:10 hrs.
At this time, approximately 2 L of rhodamine tracer was injected into well 2A and the pump was started in well 2D.

Well sampling was scheduled every 30 minutes during the tracer test. In addition, a pretest or background sample was collected from each sample location 20 minutes prior to the start of the test. At Site 5, sample sites included the canal, the injection well, wells 5D and 5A. At site 2, sample locations included the canal, the injection well (2A), wells 2B, 2C, and 2D. At both sites samples were collected for AMD and rhodamine analyses. Samples collected for tracer analyses were collected in 250-mL polypropylene bottles and stored in coolers. To avoid cross-contamination between the wells, dedicated sample tubing was installed in each well. Sample tubing consisted of a length of plastic tubing, a connector, and a short section of peristaltic tubing. The length of tubing installed in each well ensured that samples would be collected from the same elevation. For each sample collected, sufficient water was pumped to ensure the tubing was purged and the sample represented water that was in the well at that time. Sampling was conducted at both sites until 17:45 hrs, when a hail storm abruptly ended the experiment. The following morning, samples were collected from each sample location and another set of samples was collected on the morning of August 8, 2007.

3.3 Aquatic Studies

Several experiments were initiated to investigate the effect of LA-PAM on benthic and drifting macroinvertebrate communities in canals, receiving waters, and smaller-scale experimental troughs. Based on the results of the initial experiment at SD (Section 3.3.1) and in consultation with the PRP, it was determined that future studies should investigate the potential impact of LA-PAM on native BMIs in return waters rather than focusing on transient communities in canals that are dewatered yearly. An experimental trough (Section 3.3.2) was inoculated with BMI communities obtained nearby from the Colorado River. Two additional field studies were also initiated, but were terminated during the experiments. The first study utilized artificial BMI substrates in the Lost River near Klamath Falls, OR, however, poor water quality conditions in the river itself were not conducive to the growth of sensitive BMI species, resulting in the termination of the experiment. The second field study in the South Canal (Section 2.3.9) resulted in the full utilization of LA-PAM by suspended sediment resulting in no measureable concentration of the polymer in canal or receiving waters.

3.3.1 Field Study: Smith Ditch No. 7

The initial experiment was conducted along a 4-km (2.5-mi) reach of canal that extended from Kannah Creek (sourcewater) to a diversion box where water was directed into several irrigation systems. This reach was equally divided into control, treatment, and receiving reaches that were each approximately 1.3 km (0.85 mi) long. The control section was upstream from LA-PAM application, LA-PAM was applied to the treatment reach, and treated water flowed through the receiving reach (see Section 2.3.8). The SD is a small, turbid canal with fine substrates. During the experiment, flow in the canal was approximately 0.1 m$^3$ s$^{-1}$ (5 ft$^3$ s$^{-1}$). Information collected during LA-PAM transport studies found that particles move through the canal approximately 270 m hr$^{-1}$. The canal is bordered by willow (*Salix* sp.) and cattails (*Typha* sp.) and its gradient is less than 0.04 percent.
Aquatic macroinvertebrates were collected at 10 sites along SD. Sites 1 and 2 were upstream from LA-PAM application (control sites), sites 3 and 4 were within the treated reach, and sites 5 through 10 were downstream from the treated reach, but they received LA-PAM that was applied upstream. Sites 1 and 2 were 560 m and 710 m upstream from the treated reach, respectively, and sites 5 through 10 were 0 m, 100 m, 250 m, 500 m, 970 m, and 1,440 m downstream from the lowest boundary of the treatment reach, respectively. The macroinvertebrate response to LA-PAM application was examined by comparing substrate and drift communities before, after, and during application. Studies included substrate and drift sampling at control sites (1 and 2), treatment (3 and 4), and two receiving sites (8 and 9) during the afternoon for the three days prior to and following LA-PAM treatment on August 25, 2006. These collections were made by compositing six 0.11-m² (1-ft²) quadrats that were sampled by roiling and scrubbing the substrate by hand to dislodge organisms into a 250-μm, hand-held, D-frame net, with a 30-cm opening width. During each sample, quadrats were placed along a transect that crossed the wetted width and was approximately 0.50 m (1.5 ft) upstream from the transect where the previous sample was collected. The structure of BMI communities was determined by identifying a minimum of 300 randomly sampled individuals and a rare-large search (Vinson and Hawkins, 1996). This sampling method is inadequate to identify all taxa in a community, but it adequately quantifies dominant members of the community to assess changes in structure that may result from LA-PAM exposure.

Immediately prior to substrate sampling, drift samples were collected over 30 minutes in a 250-μ mesh plankton net with a 30 cm x 30 cm (12 in x 12 in) opening and a cod end with removable cap. Nets were placed approximately 3.8 cm (1.5 in) above the substrate, secured into position with vertical bars.

During LA-PAM application, drift samples were collected from control sites 1 and 2 and sites 6 through 10 in the receiving reach (site 5 was not sampled because of manpower limitations). Drift collection methods followed techniques described above but these samples were collected hourly from the time of first LA-PAM application (10 hours) for control sites and every 30 minutes for more than eight hours at most sites in the receiving reach (sites 6 through 10). The number of drift samples taken at sites 1, 2, 6, 7, 8, 9, and 10 were 16, 10, 17, 16, 16, 19, and 19, respectively. Samples from sites 6 through 10 included a number of collections before chemical analysis showed LA-PAM reached a site and following its downstream passage when LA-PAM concentrations were zero. Samples collected before the arrival of LA-PAM were pre-treatment ‘controls’ and all subsequent samples treatment collections.

All macroinvertebrates were collected alive, preserved in the field with 90 percent ethyl alcohol, and returned to the lab for sorting, identification, enumeration, and archiving. Insects were identified to the lowest possible taxonomic level, and to the lowest reasonable level for noninsect taxa. Insects were generally keyed to genus and, when possible, to lower levels, including midges (Chironomidae) to least genus, and often to species. Nondistinct taxa (e.g., organisms too damaged or small to determine with certainty) were not included in the analyses. All samples are archived in the DRI Aquatic Ecology Laboratory.
3.3.2 Controlled Trough Experiments

Effects of LA-PAM applied at 44.8 kg ha$^{-1}$ (40 lbs ac$^{-1}$) were examined in an experimental array of three replicated, untreated (control), and treatment troughs that measured 3 m (10 ft) x 10 cm (4 in) and were filled with 3.5 L (4 qt) of 1.5 cm (0.75 in) washed gravel from a quarry along the lower Gunnison River. Troughs were placed adjacent to the Grand Valley Canal in Grand Junction, CO, and supplied with unfiltered water pumped from the canal. Pumping began in mid-April, with a continuous flow of 7 L min$^{-1}$ (1.5 gal min$^{-1}$) that was maintained in each trough throughout the experiments. The slope of all troughs was the same. Periphyton was allowed to establish six weeks before the first BMI inoculation was made of BMIs from a nearby riffle in the Colorado River. A second inoculation occurred approximately eight weeks later, which was approximately four weeks before treatment that occurred in late July.

During the treatment, LA-PAM was mixed with distilled water for each treatment trough and allowed to hydrate for 30 minutes in a 1-L container. Linear anionic PAM, and distilled water for control troughs, was applied to each trough by simultaneously inverting and draining the bottles over a three-minute period. Treatment effects on benthic communities were assessed by removing substrate from the downstream-most 30 cm (1 ft) of each trough 24 hours before LA-PAM treatment. The remainder of substrate was removed 48 hours after treatment to allow BMIs time to respond to the presence of LA-PAM. Drift samples were collected before pre-treatment substrate samples were removed and during LA-PAM application using a 10 cm (4 in) x 12 cm (5 in) 250-μ mesh net held at the bottom of each trough for 30 minutes. Methods to assess BMI and drift communities followed SD studies.

3.3.3 Analytic Methods

A Hilsenhoff biotic index (HBI) was calculated for all substrate communities (Hilsenhoff, 1987). All parametric and nonparametric analyses were conducted using Systat® (Systat Software, Inc., Chicago, IL). All data were tested for normality using a Kolmogorov-Smirnov test, which showed that density, species richness, and drift rate distributions all differed significantly from normal ($p < 0.05$). The composition of pre- and post-treatment benthic communities in SD was also examined with Kendall Tau-$B$ coefficient, which evaluates similarity by calculating values between 0 and 1. Values greater than 0.75 characterize similarity, while lower values characterize dissimilarity. The effect of LA-PAM application on macroinvertebrate drift was examined for each site by comparing rates and species richness in samples collected before analytical analysis recorded LA-PAM with samples collected following application.

Relationships between BMI and drift community structure, LA-PAM and SSC concentrations, and physical characteristics of SD were also examined using canonical correspondence analysis (CCA). This gradient analysis determines environmental variables that most influence biotic community structure (Jongman et al., 1987; ter Braak and Prentice, 1988; Palmer, 1993). Axis scores were standardized using methods of Hill (1979), scaled to optimize the representation of sites, and Monte Carlo simulation (1,000 iterations) tested the hypothesis that there was no relationship between species and environmental matrices. Detrended correspondence analysis (DCA) was also used to compare the structure of pre- and post-treatment macroinvertebrate substrate and drift communities in the experimental...
troughs. For this analysis, data were detrended by segments, rare species down-weighted, and species data were log transformed. Both analyses were conducted using CANOCO v. 4.5 (ter Braak and Šmilauer, 1998). Both analyses used density estimates and only species comprising more than 10 percent of any sample.

### 3.4 Water Quality Monitoring

Water quality was monitored via field measurements of continuous turbidity and suspended sediment concentration. Samples were collected for the analysis of concentrations of suspended sediments, LA-PAM, AMD, and major ions.

Suspended sediment levels in the canals were assessed a variety of ways depending on the specific goals of the study. Grab samples for SSC were collected at nearly all sites and shipped back to DRI for SSC and laser particle size analysis (LPSA) (“G” in Table 2-1). For studies that assessed the LA-PAM concentration in the canals, SSC samples were collected in conjunction with 40 to 70 percent of the LA-PAM samples. For studies that assessed in-canal turbidity changes with LA-PAM application, water samples were analyzed in the field using a LaMotte 2020e portable turbidimeter (“TP” Table 2-1) within 48 hours of collection. A subset of these samples was returned to DRI for SSC and LPSA analyses. The Soil Characterization Laboratory (SCL) at DRI analyzed SSC following the American Society for Testing and Materials (ASTM) D 3977-97 for all samples submitted. When enough sediment weight was present in the sample, it was subsequently analyzed for LPSA using a Saturn 5200 Digisizer (Micromeritics, Norcross, GA), using a modified methodology based on ASTM C 1070-01.

For several studies, continuous turbidity was measured ranging from a few days before and after LA-PAM application up to an entire field season. Short-term deployments were used to better understand flocculation dynamics as LA-PAM interacted with suspended sediment, whereas long-term deployments were used to assess how seepage rates could vary in response to suspended sediment loads. Continuous turbidity was measured utilizing a DTS-12 (Forest Technology Systems, Blaine, WA) sensor, noted as “TA” in Table 2-1. Pressure transducers (KPSI, Hampton, VA) were also installed at a subset of these sites. For short-term deployments, the sensors were mounted vertically on a fence post installed in the canal. For long-term deployments, the DTS-12 (horizontal orientation) and pressure transducer (vertical) were mounted to a steel plate that was then lowered into the canal.

Water samples for other constituents were collected prior to most LA-PAM applications. These samples were analyzed for electrical conductivity, pH, bicarbonate, calcium, sodium, magnesium, turbidity, total dissolved solids, total suspended solids, dissolved organic carbon, and iron at DRI’s Analytical Chemistry Laboratory.

The number of LA-PAM samples varied at each site depending on goals of the specific study. During earlier studies, multiple observation points downstream of the LA-PAM application (“M” in Table 2-1) were used to assess downstream transport. Later studies collected a greater density of samples at a single downstream site (“1” in Table 2-1). These studies typically collected approximately 35 LA-PAM samples per treatment site with an additional eight samples upstream in the control. Samples were collected at a greater density near the leading and trailing edges of the LA-PAM-treated canal water and continued for up to 48 hours.
Samples were initially collected in gallon-sized water containers to minimize the impact that the inadvertent collection of partially hydrated LA-PAM granules had on the sample result. This was done by submerging the plastic jug in the thalweg of the canal and moving it up and down until it was three-quarters filled. Later, water samples were collected in 950-mL glass jars directly for grab samples or using a US DH-81 hand sampler (Federal Interagency Sedimentation Project, Vicksburg, MS) at five to seven evenly spaced spots along the canal width. Where bridges were unavailable, a single depth-integrated sample was taken about six feet from the bank with the DH-81 hand sampler.

After collection, LA-PAM samples were allowed to hydrate at ambient temperature out of direct sunlight for at least 12 hours. They were subsequently shaken for 30 seconds and decanted into a labeled centrifuge tube. Each sample was spun in a Sorvall GLC-1 (Thermo Fisher Scientific, Waltham, MA) centrifuge for at least 20 minutes at maximum speed (32,000 rev min\(^{-1}\)). The supernatant was transferred to a fresh centrifuge tube and shipped to the University of Nevada, Reno (UNR) for analysis. In the laboratory, LA-PAM concentration was determined by gel permeation size exclusion liquid chromatography using a 0.05-M potassium phosphate mobile phase, with a reporting limit of 0.1 mg L\(^{-1}\).

Water samples for AMD were either collected in a 250-mL plastic bottle for grab samples, or transferred into the 250-mL plastic bottle if a DH-81 sampler was used. These samples were immediately placed in a cooler containing dry ice and shipped frozen for laboratory analysis at UNR. The concentration of AMD was determined by gas chromatography and mass spectrometry after bromination of the water sample with a reporting limit of 0.1 \(\mu\)g L\(^{-1}\).

3.5 Laboratory Studies

3.5.1 Temperature Effect on LA-PAM Hydration and AMD Release

Laboratory experiments of LA-PAM hydration used a vibratory viscometer (model SV-10, A&D Engineering, Inc., Milpitas, CA) to measure changes in solution viscosity for LA-PAM in de-ionized (DI) water and in 0.005-M calcium sulfate solutions. A 1.7-L water-jacketed glass vessel was connected to a circulating temperature-controlled water bath. In preparation for each measurement, 500 mL of DI water was added to the vessel and, in the case for DI water only, the viscometer probe was immediately lowered into the water according to the manufacturer’s instructions. For the runs in calcium sulfate solutions, about 0.5 g of the dihydrate salt was added to the 500-mL DI water and the salt was allowed to dissolve before the viscometer probe was lowered into the solution. A large magnetic stirrer at the lowest setting was used to agitate and mix the solutions. After the water equilibrated at the preset temperatures, a measured amount of LA-PAM (approximately 7 to 15 mg, Tack Dry, Precision Polymer Corp., Greeley, CO) was added and viscosity measurements were begun.

For the DI water solutions, viscosity measurements were made at 4.6, 14.9, 15.1, 23.3, 25.4, 25.5, 30.3, and 31.1\(^\circ\)C. For the 0.005-M calcium sulfate solutions, viscosity measurements were made at 4.8, 5.0, 15.0, 15.2, 21.8, 22.3, 25.4, 30.8, and 31.2\(^\circ\)C. Viscosity and temperature were measured every five seconds and the output from the viscometer was downloaded to a laptop computer containing data processing software provided by the manufacturer. The output was displayed graphically in real time showing viscosity and temperature versus time. The solutions were monitored until the positive slopes of the
increasing viscosity measurements became near zero (slightly +/-), indicating that viscosity was no longer changing. Measurements were continued after this for at least 20 minutes to establish a viscosity plateau. After each run, the glass vessel was emptied and rinsed in preparation for the next run.

To study AMD release, about 10 mg of LA-PAM were added to 50 mL DI water in separate containers and the samples were agitated on a rotary shaker. Samples were removed after 5, 10, 20, and 40 minutes of agitation and filtered through 9-cm-diameter glass fiber filters to remove incompletely hydrated LA-PAM. The filtrates were subsampled (approximately 2 mL) and filtered again through 13-mm-diameter 0.45-micron filters into 2-mL amber autosampler vials, which were sealed with crimped aluminum caps containing Teflon-lined butyl rubber septa. The amount of AMD in each sample was assayed by high performance liquid chromatography (Agilent Model 1100) using a 300 mm x 7.8 mm Aminex HPX-87H ion exclusion column (Bio-Rad, Hercules, CA) and a diode array detector set at 195 nm (10 nm band width, 400 nm reference). The AMD was eluted using a 0.01-M aqueous sulfuric acid mobile phase at a flow rate of 1 mL min⁻¹; the column was heated to 60°C. The instrument was calibrated using standard solutions consisting of electrophoresis-grade AMD (99.9%; FisherBiotech) in DI water (1.1 to 27.3 mg L⁻¹).

3.5.2 Jar Tester Studies

Jar test studies were used to investigate the role of total dissolved solids (TDS) concentration and composition, LA-PAM concentration, and SSC on the ability to form LA-PAM-sediment flocs, an important mechanism for seepage reduction. All jar-test experiments were conducted with a CLM4 model jar tester (EC Engineering, Edmonton, Canada). For the LA-PAM and SSC Matrix experiment, various concentrations of kaolinite (0, 75, 150, 300, 600 mg L⁻¹) were dispersed in 1,100 mL of DI with 0.001-M calcium sulfate to form a sediment suspension. Granular LA-PAM was added at the pump-off stage to produce several levels of LA-PAM concentrations (0, 1, 2, 4, 8, 16, 24 mg L⁻¹). Experiments were run in random order to prevent cross-contamination of equipment or bias. Some combinations were run in triplicate for quality assurance.

For the sodium adsorption ratio (SAR) and TDS Matrix experiment, fixed amounts of kaolinite (225 mg L⁻¹) and LA-PAM (1.5 mg L⁻¹) were added to each cell. The TDS concentration in the bulk solution was varied (100, 600, 1,100, 1,600 mg L⁻¹), as was the ratio of sodium to calcium in the solution. The ratio of sodium to calcium was determined by the SAR (0, 0.1, 1, 3, 6). To achieve these combinations of TDS and SAR, various levels of the salts (CaSO₄, Ca(NO₃)₂, MgSO₄, and NaNO₃) were utilized. The ranges of TDS and SAR were determined from concentrations commonly found during LA-PAM application field studies.

Each of the four cells of the jar tester were plumbed with tygon tubing from the outlet at the mid-point of the paddles and connected to an in-line turbidimeter for the first experimental matrix, LA-PAM versus SSC. A Cole-Parmer Masterflex (model 7519-06, Vernon Hills, IL) peristaltic pump pulled water sample to the four parallel turbidimeters. Three MicroTol4 (model 20063, HF Scientific, Fort Meyers, FL) flow-through turbidimeters were employed with a Hach model 2100AN (Loveland, CO) turbidimeter for the fourth jar tester cell. For the second experimental matrix, SAR versus TDS, the sample was pumped from tubing attached to the baffle just above the paddles and the Hach turbidimeter was
replaced with a fourth MicroTol4 for measurement consistency. Return flow from the turbidimeters to the cells went through an inverted enlarged fitting, which allowed particles to settle easily. Flocculation procedures were modified from well-known industrial standard methods (Hudson and Wagner, 1981; U.S. EPA, 1999). First, a background high-paddle-speed kaolinite and salt hydration period of 12 minutes was completed. Second, the peristaltic pump was turned off for 60 seconds while granular LA-PAM was added to the jar tester cells. This allowed LA-PAM to mix and hydrate in the salt/kaolinite solution without being pulled into the tubing and forming globules that may have adhered to the tubing. Third, the pump was turned back on and paddles were run at high speed (100 revolutions min⁻¹) for four minutes, where the floculant and sediment were spun together during the flash stage. Fourth, the paddle speed was lowered to approximately 15 revolutions min⁻¹ for 25 minutes to facilitate flocculation. Finally, the mixing paddles were shut off and followed by an additional 20-minute monitoring period.

3.5.3. Potential Transport of AMD in Groundwater

The fate and transport of AMD in soil/water systems was studied in the laboratory (Arrowood, 2007). Batch experiments were initially run to understand the sorption characteristics of AMD in the same three soil materials utilized in previous studies (Young et al., 2007b, c). An unbalanced factorial experimental design was used with four reactant (i.e., soil) materials, five AMD concentrations, five mixing periods, and two soil pretreatments (autoclaving) following the methods of Papiernik and Yates (2002) and Arrowood (2007). All supernatant samples were analyzed by high-performance liquid chromatography with a method detection limit of approximately 3 μg L⁻¹.

Column experiments were also conducted to measure breakthrough of a bromide tracer and AMD in sterile soil. These experiments utilized an unbalanced 2-factor design with three soil types, and five AMD concentrations included as independent variables. Each combination was run in duplicate using the experimental method described by Paperniek and Yates (2002). The column experiments were re-executed to examine how microbial breakdown of AMD affected the breakthrough curves. Four identical tests were also run with soil columns filled with undisturbed soil from the Rocky Ford Highline Canal, CO.

Transport results from the laboratory experiments were subsequently used as input to HYDRUS-2D (Šimůnek et al., 2006) to simulate the fate and transport of AMD from a canal prism and into groundwater. HYDRUS-2D is a numerical code that simulates water flow and solute transport in variably saturated media. The conceptual model included an symmetric flow domain with a canal prism as the source of water and AMD. The canal prism was represented as either: 1) untreated; 2) partially sealed with LA-PAM; or 3) fully sealed with LA-PAM. Soil underneath the canal and throughout the flow domain was represented as one of the three soil types used in the laboratory experiments (i.e., #70 mesh sand, C33 sand, or loamy sand soil) using hydraulic properties quantified in the laboratory. Acrylamide was then introduced into the flow field at the canal prism/soil boundary and allowed to migrate toward a water table positioned at three depths (at base of canal, 1 m below canal, and 10 m below canal). Observation nodes were placed at various locations in the flow field, ranging in distance from 0.92 to 25.04 m from the inlet point for AMD, so that concentrations of AMD could be estimated with time. The model was run for 50 days with an AMD pulse length of five days, or about five times longer than would be expected from a typical field application of LA-PAM.
4.0 FUNDAMENTALS OF LA-PAM APPLICATION

Although the application of granular LA-PAM to water delivery canals appears to be straightforward, a variety of factors must be accounted for to maximize seepage reduction while also reducing potential environmental impacts. The most important factor is the selection of the proper PAM to apply to the canal. Different types of PAM include cationic, anionic, and nonionic cross-linked. Each type is comprised of different product formulations covering a large range of molecular weights, charge densities, environmental impacts, and reactivities in water. Nonionic cross-linked PAMs are typically not used for flocculation in aqueous systems due to their lower reactivities (Mason et al., 2005). Cationic PAMs are typically not used in open systems due to adverse impacts on aquatic animals (Goodrich et al., 1991; Muir et al., 1997). Linear anionic PAMs, however, are both reactive and appear to pose little environmental impact (Young et al., 2007a) when used responsibly. The specific granular LA-PAM formulations and application procedures developed during these studies are detailed in Section 2 and by Susfalk et al. (2007). The following section provides an overview of the chemical and physical processes thought to be involved in the flocculation of LA-PAM with suspended sediment in water delivery canals. Several management and application issues are highlighted in Section 4.3, and an example of how these factors affected flocculation under field conditions is presented in Section 4.4. This case study is intended to highlight the principles discussed here, and a more in-depth discussion covering all the sites studied is presented in Section 9.

4.1 Formation of PAM-sediment Flocs

An effective application of granular LA-PAM starts with proper application of the polymer to the water surface. The most effective approach is to use a method and rate of application that maximizes the contact of LA-PAM with the suspended solids in the water column. If granular LA-PAM is added too quickly, such as if poured directly from its shipping bag into the canal, partially hydrated LA-PAM agglomerates will form. These agglomerates do not easily dissolve and, as a result, they travel downstream, become entwined in bank foliage, circulate in eddies, and do not participate in the seepage reduction process. Within the studies described in this report, agglomerates were avoided by using manual or automated spreaders to evenly disperse the LA-PAM across the water surface.

When granular LA-PAM is exposed to water, it dissolves to release the LA-PAM polymer and interact with water molecules through hydrogen bonding and dipole interactions. Initially, PAM concentrations are locally high around the granule, with the polymer remaining coiled or in weakly extended conformations that hinder its reactivity towards suspended solids (Misra, 1996). As the polymer continues to disperse and be diluted by the canal water, it will become more extended and less coiled, thereby becoming more reactive toward the charged mineral surfaces residing on the suspended solids (Malik and Lety, 1991). The pH of the matrix solution may play a role, as LA-PAM typically assumes a more random coil conformation at low pH (e.g., pH<5), whereas electrostatic repulsion at higher pH (e.g., pH>8) results in a more extended conformation (Tjipangandjara and Somasundaran, 1992). Individual polymer strands are capable of interacting with a variety of suspended solids along the length of their chains. Flocculation occurs when an electrostatic attraction between smaller PAM-sediment particles develops a larger association called a
floc. The PAM-sediment floc continues to increase in size until it becomes heavy enough to settle to the bottom of the canal.

Several factors affect the ability of PAM-sediment flocs to form. The presence of cations promotes flocculation through charge screening (Lu et al., 2002) and cation bridging. In canal waters, negatively charged carboxylate groups on the LA-PAM chain and the negatively charged sites on sediment particles can only interact if a positively charged cation acts as a bridge between the two negatively charged sites (Theng, 1982; Nadler and Letey 1989, Letey, 1994). This reaction is called cation bridging and is thought to be the primary mechanism through which the anionic polymer and suspended solids react (Lu et al., 2002; Nabzar et al., 1984). Secondary mechanisms have been proposed through hydrogen bonding (Nabzar et al., 1984; Pefferkorn et al., 1990), ligand exchange (Theng, 1982), or hydrophobic bonding (Laird, 1997). These mechanisms appear to be less important than cation bridging, however, and are thought to occur under conditions that are atypical of canal systems.

The formation of PAM-flocs is also affected by conformation of the polymer chain, which tends to become smaller in aqueous solutions containing salts (Muller et al., 1979), and increases the likelihood of a greater number of LA-PAM molecules interacting with the sediment. Anionic polymers are more reactive when dissolved in solutions having a higher concentration of dissolved salts (Ben-Hur et al., 1992; Lu et al., 2002; Aly and Letey, 1988) and a greater composition of divalent cations. Divalent cations, like calcium, are better able to form cation bridging, whereas monovalent cations, like sodium, have a larger, hydrated radius that impairs the formation of cation bridging. Lentz and Sojka (1996) found anionic PAM to be less effective when the ratio of sodium to calcium increased from 0.7 to 9.0 \([\text{m mol L}^{-1}]^{0.5}\). However, the importance of divalent cations for floc formation may differ with soil mineralogy and texture, as anionic PAM is more reactive with soils dominated by kaolinite, compared to smectite and vermiculite (Peng and Di, 1994; Laird, 1997; McLaughlin and Bartholomew, 2007), and in finer-textured soils that have a greater charge density compared to coarser soils (Lu et al., 2002). Flocculation is also highly dependent on the physical process of mixing the polymer and sediment, with better flocculation performance in laboratory studies under conditions of continuous polymer addition over extended time periods (Hogg, 1999).

Flocculation continues if reaction sites remain open along the LA-PAM chain, even after the flocs have settled to the bottom of the canal. Only a few segments of the polymer typically react with any given sediment particle, leaving the remaining polymer chain open to form long loops and tails, providing a relatively large “grappling distance” to facilitate interparticle associations (Theng, 1982). At the low flocculant dosages typically found in canal applications, complete and irreversible adsorption appears to occur rapidly, due to the large number of adsorption sites per polymer (Nadler et al., 1992; Taylor et al., 2002). Although each individual electrostatic interaction between a charged polymer and particle site is weak, the cumulative number of interactions per polymer segment and the number of interactions along the polymer chain result in strong adsorption. However, PAM-sediment flocs can break if the strength of the forces holding the floc together is insufficient to withstand the local hydrodynamic stresses (Ray and Hogg, 1987) within the canal prism.
4.2 Impact of PAM-sediment Flocs on Seepage Reduction

Once the LA-PAM-sediment flocs have formed and settled to the bottom of the canal, three physical mechanisms are proposed to explain the reduction of seepage. First, LA-PAM-sediment flocs can plug large soil pores, especially in coarser-grained canal sediments. Second, the PAM-sediment flocs form a low-conductivity layer. Finally, water will flow more slowly through soil pores due to a higher solution viscosity caused by the presence of LA-PAM. This section presents an overview of laboratory investigations of these methods with greater detail presented in Young et al. (2007b,c) and Moran (2007).

Results of laboratory soil column experiments showed that increasing PAM concentrations from 0 to the equivalent of 44.8 kg ha\(^{-1}\) (40 lbs ac\(^{-1}\)) in the absence of suspended sediment led to decreased K\(_{sat}\) in the three soils tested (#70 mesh sand, C33 sand, loamy sand soil). For example, at the application level of 44.8 kg ha\(^{-1}\) (40 lbs ac\(^{-1}\)), K\(_{sat}\) was reduced 80 percent, 81 percent, and 52 percent for the #70 mesh sand, C33 sand and loamy sand soil, respectively (Young et al., 2007b; Moran and Young, 2007). A portion of this reduction was accounted for by viscous effects in the test solution, but it is more likely that PAM accumulated at the soil surface and either clogged larger soil pores or otherwise formed a distinct layer separate from the soil. When sediment was added to the solution, under conditions more similar to those expected in the field, PAM and sediment together were more efficient as compared with either component being tested separately. For example, when LA-PAM at 5.6 kg ha\(^{-1}\) (5 lbs ac\(^{-1}\)) was mixed with suspended sediment at 300 mg L\(^{-1}\), the final K\(_{sat}\) values were reduced by more than 90 percent. Linear anionic PAM mixed with suspended sediment led to an 11-fold reduction in K\(_{sat}\) versus PAM alone.

The treatment of soil was also evaluated in terms of efficiency, defined as the percent reduction in K\(_{sat}\) (versus untreated soil) divided by the treatment level. It was noted that treatment combinations reduced K\(_{sat}\) only to a certain degree and LA-PAM concentrations greater than 5.6 kg ha\(^{-1}\) (5 lbs ac\(^{-1}\)) did not further lower K\(_{sat}\). For example, K\(_{sat}\) reductions only increased from 94 percent to 98 percent in the #70 mesh and C33 sands with LA-PAM concentrations of 5.6 kg ha\(^{-1}\) (5 lbs ac\(^{-1}\)) and 44.8 kg ha\(^{-1}\) (40 lbs ac\(^{-1}\)), respectively. The coarser-grained soils showed greater reductions in K\(_{sat}\), as well as lower absolute values than the loamy sand soil. Without pore-plugging, the PAM-treated loamy sand soils should have K\(_{sat}\) values lower than either of the sandy-textured soils. This result was not observed in these experiments. The results showed LA-PAM was less effective in treating loamy sand soil than the coarser-grained materials tested. In soils dominated by fine-earth fractions, seepage through these soils would already be low and canals with this subsoil would not likely warrant PAM treatment.

The laboratory experiments show that K\(_{sat}\) reduction could be due to a combination of the three mechanisms listed above. When suspended sediment is present and LA-PAM flocs form, the combination of these processes further reduces the K\(_{sat}\). Results support the hypothesis that PAM can plug larger pores in coarser-grained material, in addition to creating a surface seal. The combination of results from the laboratory experiments point to the role of floc formation in LA-PAM efficiency. For example, the maximum settlement of suspended sediment during the jar tester experiments was observed when LA-PAM was mixed into the solution at 2 to 4 mg L\(^{-1}\) (Figure 2 in Young et al., 2007c), which corresponds well with the maximum efficiency of LA-PAM treatment at 5.6 kg ha\(^{-1}\) (40 lbs ac\(^{-1}\)) (Figure 2.5 in Young et al., 2007c), which also is equivalent to a concentration of 4 mg L\(^{-1}\).
4.3 Management of the PAM-sediment Floc to Maximize Seepage Reduction

Factors already discussed, such as developing a proper LA-PAM dose and understanding how physical factors affect PAM-floc formation under given environmental conditions, must be considered to manage environmental risks while maximizing seepage reduction. When targeting a specific canal reach known to have high seepage rates, consideration of the travel time between the application of the granular LA-PAM and the deposition of PAM-sediment flocs is critical for optimal efficiency. As the granular polymer is added to flowing water, LA-PAM will move downstream, dissolving, hydrating, and reacting with suspended sediment. Travel time is dependent on several factors including the suspended solids concentration in the water, water temperature, water velocity, and water chemistry. If travel time is not considered, or if LA-PAM is applied in excess of what is needed to form flocs, it is likely that LA-PAM will travel downstream beyond the target reach. This scenario is costly and wasteful in terms of LA-PAM and, more importantly, may have unintended consequences downstream. Examples of potential environmental impacts include:

1. Introduction of LA-PAM into downstream receiving waters where the potential exists for inadvertent lining or armoring of these areas, negatively impacting aquatic communities such as by sedimentation stresses to macroinvertebrates or the potential burial of spawning gravels for fish
2. Reducing a dependent source of groundwater recharge for well users
3. The use of LA-PAM-treated canal water on fruit and vegetable crops that have little washing before market
4. Reducing the water delivered to adjacent wetlands dependent on canal seepage, and
5. The potential, but currently not quantified, interaction between LA-PAM and aquatic herbicides used to treat invasive species in water delivery canals.

Environmental exposure can be minimized through application of LA-PAM at the proper dosage (e.g., not overdosing), under environmental conditions that maximize PAM-floc formation, and limiting the application of LA-PAM to specific canal reaches known to have higher seepage rates. The acute and long-term human health risk from exposure to LA-PAM appears to be minimal (Young et al., 2007a). However, proper personal protective equipment is required during application to minimize inhalation and slip hazards. Application guidelines for the addition of LA-PAM into unlined water delivery canals have been developed to specifically reduce environmental and human health exposure while still providing seepage reduction benefits (Susfalk et al., 2007).

4.4 Case Studies: Travel Time and “Clear Zone” Development

When flocculation occurs and the PAM-sediment flocs begin to settle, the concentration of suspended solids remaining in the water column is reduced. This decrease in observable turbidity is termed the “clear zone.” As flocculation and settling are not instantaneous when LA-PAM is applied to the canal, the clear zone will develop at some point during application. The extent of the development of the clear zone is dependent on both the relative concentrations of suspended sediment and reactive PAM in the water column – inadequate amount of polymer or suspended sediment results in little development of the clear zone, whereas an excessive amount of polymer will not fully react and may be
transported downstream. If enough polymer is added to react with the majority of suspended solids, however, then turbidity will drop and the water will become clear.

The development of the clear zone was explicitly studied along two sites on the HID and one site along the RFH. For these studies, the polymer was continuously applied from a stationary location rather than being applied from a platform moving upstream. This simplification resulted in the development of a stationary clear zone at all three sites. The clear zone was assessed by the analysis of water quality samples collected from sites downstream of the application (Figure 4-1). At the HID, the clear zone never developed at the closest observed points (0.8 to 0.9 km; red lines in panels 1 and 2 of Figure 4-1), indicating the settling of PAM-sediment flocs did not occur within the first 20 to 24 minutes after application. Turbidity declines were observed within 37 and 47 minutes of application (green line in panel 1 and green and blue lines in panel 2); however, these turbidity reductions were initially small relative to what was eventually achieved. Full development of the clear zone appeared between 1 and 1.3 hours at a point between 1.9 and 2.7 km downstream from application at ambient water velocities. At the RFH, PAM-sediment floc formation appeared to be faster, as a partial clear zone was established within 32 minutes and fully developed within 41 minutes of LA-PAM addition, corresponding to a travel distance of 0.9-1.3 km at ambient water velocities.

The more rapid development of the clear zone at the RFH was due to better ambient conditions that promoted the formation of PAM-sediment flocs. First, the rate of hydration of granular LA-PAM is faster in warmer waters (Section 9.1.3). Water temperatures at the RFH averaged 24.5°C, compared to 23.4°C and 21.4°C at the eastern (HID-1) and western (HID-2) reaches of the HID, respectively. Second, the average water velocity at the RFH was lower (0.53 m s⁻¹ or 1.73 ft s⁻¹) than that observed at the HID (0.61 m s⁻¹ or 2.0 ft s⁻¹ at HID-1, and 0.70 m s⁻¹ or 2.3 ft s⁻¹ at HID-2). Lower water velocities promote the more rapid settling of the PAM-sediment floc, while higher water velocities keep the same-sized particles entrained in the water column longer. Entrained particles may also promote adsorption via increased contact time between the polymer and suspended solids and by better mixing at in-canal check structures. For example, the more rapid clear zone development at HID-2 compared to HID-1 may have been due to the presence of a check structure 0.2 km downstream of LA-PAM addition and the faster water velocities, despite colder water temperatures, at HID-2. Third, the canal water at RFH had three times the concentration of calcium and magnesium than that at HID, resulting in conditions that were more favorable to the formation of cation bridging between the polymer and suspended sediment (Section 9.1.2.). Finally, there was higher turbidity in the canal at RFH: 190 NTU compared to 45 and 18 NTU at HID-1 and HID-2, respectively. When turbidity is higher, there is a greater likelihood for polymer-sediment interactions. The result is an increase in the number of sediment particles adsorbed to each individual polymer chain, as well as faster flocculation and settling, which promote removal of suspended solids from the water column and more rapid development of the clear zone (Section 9.2.2).
Figure 4-1. Development of the clear zone during the stationary LA-PAM application studies. Turbidity was reduced as PAM-sediment flocs formed and settled to the bottom of the canal.
At HID-2, LA-PAM was measured in the water column only when the clear zone was present (Figure 4-2). The clear zone is where elevated concentrations of hydrated PAM are most likely to be found. The LA-PAM concentrations observed during this study were low, and remained below the EPA drinking water limit of 1.0 mg L\(^{-1}\). The HID is a large canal (>23 m\(^3\) s\(^{-1}\) or 800 ft\(^3\) s\(^{-1}\)), but the very low turbidity levels meant that relatively little PAM was needed to floc the sediment. Since excess LA-PAM was present, its concentration did not decrease as the clear zone passed sites 1.9 and 3.6 km downstream of application. The clear zone continued downstream, as was observed at the farthest downstream station 10.5 km from the application (Figure 4-3).

![Graph showing LA-PAM concentrations and turbidity levels](image1)

**Figure 4-2.** Polymer concentrations and turbidity levels observed during the HID-2 stationary LA-PAM experiment. PAM concentrations at HID-1 and RFH were not presented, as they were below the reporting limit.

![Graph showing continuous turbidity measured downstream](image2)

**Figure 4-3.** Continuous turbidity measured downstream of LA-PAM application at HID. The drop in turbidity represents the clear zone for each of the polymer additions. The sensor was located 6.0 km downstream of HID-1 application and 10.5 km downstream of HID-2 application.
For the HID-1 and RFH experiments, the addition of LA-PAM was not in excess of the assimilative capacity of suspended sediment, as LA-PAM concentrations remained below detection limits. Clear zones were established at both sites, though the turbidity levels caused by higher suspended sediment loads were capable of reacting with all added polymer. For the HID experiments, these differences are striking, as the change in turbidity level from 18 NTU (HID-2) to 45 NTU (HID-1) resulted in the consumption of all the LA-PAM. Even if a clear zone does not fully develop, enough flocculation may occur at higher turbidity levels to still have a seepage reduction effect. However, the clear zone may not develop in situations where the water chemistry does not promote PAM-suspended sediment flocculation. These results indicate that potential downstream impacts are identified prior to LA-PAM application. The best approach to minimize LA-PAM concentrations in the water column and reduce potential downstream risks is to apply LA-PAM during higher suspended sediment loads that are capable of fully reacting with the mass of polymer applied to the canal.
5.0 BENEFITS OF LA-PAM APPLICATION I: SEEPAGE REDUCTION

The use of seepage reduction technologies can have a substantial significance economically, as a water conservation tool, and as a method of improving surface and groundwater quality. Compared with conventional channel lining materials, LA-PAM is inexpensive, easy to apply, and has a flexibility of application. In the Grand Valley of western Colorado, the seepage reduction from canals is a critical component to reducing the deep percolation of irrigation water, which mobilizes salt and selenium loads to the Colorado River. In the LARV of eastern Colorado, reduced canal seepage would lower the water table under irrigated agricultural fields, enhancing river water quality by reducing evapoconcentration of salts associated with upflux to nonbeneficial ET on noncultivated fields, and by lessening subsurface flows that mobilize salt and selenium loads to the river (Gates et al., 2006, Mueller-Price and Gates, 2008). Benefits gained from salvaged water, however, may not always be clear-cut. Water law and the Arkansas River Compact with Kansas prohibit reduced seepage volumes in the LARV from being entirely converted to beneficial use by the canal company and shareholders. It is possible that some portion of the reduced seepage volume resulting in a reduction in nonbeneficial upflux may be able to be salvaged and converted to beneficial crop consumptive use. Decreased seepage may not always be considered a benefit, however, and would adversely affect adjacent wetlands and areas that depend solely on groundwater that is significantly recharged by canal seepage.

Seepage losses from canals are undoubtedly significant but are difficult to quantify on a reach-basis; even if a value can be estimated, there are uncertainties in scaling to canal-system and regional levels. During this study, several canal reaches in Colorado were frequently assessed for seepage rates. These data, combined with historical seepage studies conducted by Colorado State University at numerous canal reaches in the LARV of southeastern Colorado are presented in Table 5-1. Typical seepage rates ranged from 0.01 to 0.06 m³ s⁻¹ km⁻¹ and represent a loss of between 0.1 and 2.5 percent of water per kilometer of canal.

Results presented below show that: 1) where LA-PAM was effective (8 of 11 experiments), seepage rates measured within 24 hours were reduced between 28 and 87 percent; 2) based on measurements from four sites, seepage reduction was effective throughout the rest of the irrigation season; 3) variability in seepage rates over time was lower after LA-PAM treatment than before treatment; 4) assuming a conservative 35-percent reduction in seepage, LA-PAM application to the study reaches on the RFH and the FL would provide an estimated seepage reduction water savings of 1.6 x 10⁶ m³ and 8.5 x 10⁶ m³ on a yearly basis, respectively; 5) scaled up to the entire canal, this 35-percent seepage reduction could result in approximate salvaged water volumes of 6.7 x 10⁶ m³ and 23.8 x 10⁶ m³ at the RFH and FL, respectively; 6) seepage reduction benefits were not completely maintained through the winter when the canals were dewatered; and 7) inadequate environmental conditions, such as low SSC, and errors in trial application methodologies were responsible for low seepage reduction in three of the 11 LA-PAM experiments.
Table 5-1. Seepage rates from unlined water delivery canals. Data were collected as part of this study except for 2001 to 2005 LARV data provided by CSU.

<table>
<thead>
<tr>
<th>Canal</th>
<th>Water Year</th>
<th>Number of Reaches</th>
<th>Typical Q m$^3$ s$^{-1}$</th>
<th>Typical Canal Seepage without PAM Application m$^3$ s$^{-1}$ km$^{-1}$</th>
<th>Wetted Perimeter m</th>
</tr>
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<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Min Ave Max</td>
<td>Min Ave Max</td>
<td></td>
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<tr>
<td><strong>Grand Valley, Western Colorado</strong></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Smith Ditch (SD)</td>
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<td>0.01 0.02 0.02</td>
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<tr>
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<td></td>
<td></td>
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<td>Amity</td>
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<tr>
<td></td>
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<td>4.3</td>
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<td></td>
<td>0.04 0.04 0.04</td>
<td>15.8</td>
</tr>
</tbody>
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*Seepage measurements were taken during specified intervals for a duration prior to PAM application.

5.1. Seepage Rate Reduction in Study Canals

In the LARV, seepage reduction was estimated from volume balance studies and was evaluated both in the short term (within the first month after LA-PAM application) and long term (within three to five months after application). In western Colorado, seepage rate estimates were based on the volume of water entering and exiting the treatment reach through flumes while accounting for the travel time of canal water. For data collected in 2005 and 2006, emphasis was primarily on short-term seepage reduction measured during the 24 hours...
before and after LA-PAM application. In 2007, emphasis was placed on the collection of longer-term datasets at a small number of canal reaches, warranting more complex seepage estimation methods that are discussed in Section 3.1.

5.1.1. Short-term Seepage Reduction

Initial LA-PAM application studies were conducted in the lower volume canals of western Colorado in 2005 and 2006 (Table 5-2). These experiments provided the opportunity to refine both the LA-PAM application and sampling methodology prior to studies on larger canals. Seepage reduction was 28 to 85 percent in these canals, but difficult to measure because the canals were small, the absolute amount of water lost was low, and small change in flow destabilized discharge measurements, causing a larger perturbation in seepage estimates. For example, flow in KC doubled within hours, depending on the need of a single downstream user. At MD, offtakes remained open during the study, and the flow entering the canal was reduced by half the morning that LA-PAM was applied. The presence of active, underground pipe offtakes prevented accurate estimation of actual seepage rates. However, because diverted flows were held constant during the experiment, percent seepage reduction could be estimated. For the KC-2 experiment, the 28 percent reduction in seepage rate after LA-PAM application, coupled with the already high water volumes, threatened to spill water over the canal bank. Within 24 hours, the canal company partially breached the canal and dumped the salvaged water into a nearby creek to prevent an unwanted breach from occurring further downstream.
In the LARV, seepage rate reductions after LA-PAM application ranged from -49 to 87 percent. Seepage rates were not reduced in two of the six experiments; LAM-1 and RFH-1. At LAM-1, suspended sediment levels were extremely low, requiring the addition of sediment for LA-PAM to form flocs. Sediment was added by raking the sides of the canal upstream of the LA-PAM application as discussed in Section 2.3.5, but the effort was ultimately unsuccessful as more fully addressed in Section 9.2.2.

At LAM-1, the measured increase in seepage could have been due to three factors. First, low discharge coupled with naturally low seepage rates within this reach resulted in discharge measurements that were within each other’s error bounds. Second, the raking of the canal and the disturbance by the work crew walking the canal may have disturbed an existing low conductivity surface layer, thereby increasing seepage. Third, the presence of low concentrations of LA-PAM in the water column without the formation of PAM-sediment flocs could have stabilized soil structure in the canal bed, leading to an increase in seepage. This experiment would likely have been successful if the application of LA-PAM was delayed until sufficient suspended sediment was present in the water column, as discussed in the LA-PAM application guidelines (Susfalk et al., 2007). Under these conditions, the best strategy may be to apply LA-PAM immediately following a thunderstorm, when SSC is elevated.

The lack of observable seepage reduction during the RFH-1 experiment stemmed from the very low seepage rate per km of canal and the low SSC. These issues were rectified during the RFH-2 experiment the following month, when LA-PAM caused a 59-percent reduction in seepage in a shorter reach that had a greater relative seepage rate and four times higher SSC. The application of LA-PAM reduced seepage by 39 to 87 percent in the four experiments that had the highest suspended sediment levels (471 to 1745 mg SSC L⁻¹).

Finally, three factors may explain why an LA-PAM application did not alter seepage rates in the HID. First, although PAM-sediment flocs formed and created a clear zone (Section 4.4), sediment levels may have been too low to affect seepage rates. Second, this was a stationary LA-PAM application (Sections 2.1.3 and 2.3.3), and the clear zone was only observed at 0.8 to 2.7 km downstream of the application point, so the canal lengths before and after this zone were essentially untreated. Finally, the large flows and open diversions within the treatment reach increased the error in seepage measurements and small differences may have been missed.

5.1.2 Long-term Seepage Reduction

Seepage rates were estimated several times up to five months after LA-PAM addition to the CAT, LAM, and RFH canals in 2006 and the CAT in 2007. In all four cases, LA-PAM continued to be effective at reducing seepage throughout the rest of the irrigation season (Figure 5-1, Table 5-3). It is unknown why temporary increases in seepage, apparently unrelated to changes in turbidity or stage, occurred during the first month at three of the sites.
Figure 5-1. Seepage rates immediately prior to and up to five months after the application of LA-PAM on canals in the LARV. Dotted lines connecting the points are included for visual clarity and do not represent estimated seepage rates between measured points. The vertical red line indicates when LA-PAM was applied.

Table 5-3. Estimated seepage reduction as percentage. Values based on seepage rate the day prior to LA-PAM application. Time periods in which no measurements were conducted are denoted as ‘NA’.

<table>
<thead>
<tr>
<th>Elapsed Time (days)</th>
<th>LAM-2</th>
<th>CAT-1</th>
<th>CAT-2</th>
<th>RFH-2</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>33</td>
<td>99</td>
<td>100</td>
<td>78</td>
</tr>
<tr>
<td>2 to 15</td>
<td>27</td>
<td>NA</td>
<td>82</td>
<td>64</td>
</tr>
<tr>
<td>16 to 55</td>
<td>31</td>
<td>90</td>
<td>91</td>
<td>71</td>
</tr>
<tr>
<td>56 to 115</td>
<td>33</td>
<td>89</td>
<td>100</td>
<td>67</td>
</tr>
<tr>
<td>&gt;116</td>
<td>NA</td>
<td>93</td>
<td>NA</td>
<td>NA</td>
</tr>
</tbody>
</table>

The impact of LA-PAM application on groundwater levels was investigated for the CAT-2 experiment using several monitoring wells installed along the canal. Groundwater levels in the wells closest to the canal declined as a result of seepage reduction but levels further from the canal were not as evident (Appendix). Groundwater levels continued to decline by a small amount for one to two months after PAM application, however, this decline could have been attributed to decreased canal flow rates and canal water surface levels that occurred later in the water year.

5.1.3 Seepage Estimates using Temperature as a Tracer

Thermocouple data from the Highway 50 site at the RFH near Fowler, CO, were preliminarily investigated. This section of canal was comprised of fine, silty sediments, with
steep canal geometry and an average water depth of 1.1 m (3.5 ft) for the majority of the 2007 measurements. Data from three periods have been examined to date: 1) before LA-PAM application (May/June 2006); 2) after LA-PAM application (July 2006); and 3) after a thunderstorm that produced extremely high sediment loads that “self-sealed” the canal (August 2007).

Data collected before LA-PAM treatment showed a distinct pattern of lessening amplitude, longer periods, and lagged maximums between the water and the upper two thermocouples (Figure 5-2). This pattern was absent from the lower three probes, for which the daily temperature maximums were aligned and the amplitude did not change. This indicates that water moved more slowly in the top 0.3 m (1 ft) of sediment, which was therefore the confining layer that determined the amount of water lost to groundwater. It is possible that water moved laterally toward the banks and joined regional groundwater movement at the lower depths, which might explain the difference between the patterns in the top two and bottom three probes. The lag time of 14 to 18 hours between temperature maxima in TC1 and TC2, which are separated by 0.15 m (0.5 ft), indicates a seepage rate of 0.2 to 0.3 m day$^{-1}$ (0.7 to 0.9 ft day$^{-1}$). Temperatures modeled with HYDRUS showed a good fit to observed data ($r^2$ = of 0.97) (Figure 5-3a) and suggest a seepage rate of approximately 0.1 m day$^{-1}$ (0.3 ft day$^{-1}$ or 0.2 ft$^3$ s$^{-1}$ ac$^{-1}$) through this top layer. These estimates are consistent with the 0.1 to 0.3 m day$^{-1}$ (0.2-0.8 ft day$^{-1}$ or 0.1-0.4 ft$^3$ s$^{-1}$ ac$^{-1}$) calculated by StreamPro measurements during summer 2006.

![Figure 5-2. Selected thermocouple data from the water column, 0.15 m (0.5 ft) below the sediment surface (TC1) and 0.30 m (1.0 ft) below the sediment surface (TC2) collected before (left) and after (right) PAM application in 2006. Modeling of thermocouple data involves analysis of amplitude (A), period (B), and lag time (the difference in time between temperature peaks at different depths) (C), to determine hydraulic conductivity. Changes in these properties help to determine differences in seepage over time.](image-url)
Figure 5-3. HYDRUS results for three thermocouples located at depths of .15, 0.30, and 0.60 m (0.5, 1.0 and 2.0 ft) below the sediment in May/June 2006 before PAM was applied (a), July 2006 after application of PAM (b), and during the “self-sealed” period in August 2007 (c). The solid lines represent modeled temperatures, while the points are the observed thermocouple data.
LA-PAM was applied along this reach on June 30, 2006. Surface water discharge measurements collected immediately after application did not show a significant decrease in seepage along this six-mile reach. Changes in the seepage rate would be reflected in the thermal profile by a change in daily temperature amplitude, i.e., if less water was flowing into the sediment, the daily temperature fluctuations are reduced at depth. Running daily mean temperature was calculated for the two-week period before and after PAM application. Fluctuations about this mean temperature did not show a difference in amplitude before and after PAM application, suggesting that the PAM application was not effective at this location (Figure 5-4). Temperatures modeled with HYDRUS (Figure 5-3b) fit the observed thermocouple temperatures reasonably well ($r^2 = 0.69$) and suggested a decrease in saturated hydraulic conductivity to 0.04 m day$^{-1}$ (0.14 ft day$^{-1}$ or 0.07 ft$^{3}$ s$^{-1}$ ac$^{-1}$). The poor fit between simulated and measured temperatures was partly due to the introduction of noise in the measured temperature data due to issues with the thermocouple cold-junction compensation.

A second LA-PAM application was planned for this site in August 2007 based on seepage estimates from July 2007; however, a large thunderstorm at the beginning of August mobilized a high concentration of fine sediment that naturally sealed the canal. The daily fluctuations about the mean temperature during this time period did show a slight but consistent decrease in amplitude, implying reduced seepage (Figure 5-5). Temperatures modeled with HYDRUS (Figure 5-3c) fit the observed data reasonably well ($r^2 = 0.76$) and suggested a decrease in saturated hydraulic conductivity to approximately 0.1 m day$^{-1}$ (0.2 ft day$^{-1}$ or 0.1 ft$^{3}$ s$^{-1}$ ac$^{-1}$). Seepage rates from the thermal tracer approach could not be directly compared with the surface water measurement approach due to changes in surface water discharge measurement locations, and open diversions along the reach during measurements. However, surface water discharge measurements upstream of this site recorded a sudden drop in seepage during this time. Analysis of data from four other sites may show greater changes in the seepage rates. The relatively small differences in hydraulic conductivity observed at this site result from the existing soil hydraulic properties and
possibly due to the silty sediment that has been shown to have low PAM effectiveness (Young et al., 2007b).

An important aspect of canal seepage that cannot be addressed by discharge measurements is heterogeneity within a canal reach. A better understanding of differences within the canal bed would provide valuable information about specifically what areas of the canal are losing the most water, and where to direct future efforts to prevent water loss. Ponding tests can help provide these data but are difficult to carry out with most canal operations during the irrigation season.

Initial studies using a newly published technique were conducted in 2008 to gain additional insight on canal bed heterogeneity. This method required monitoring the wetting front velocity when the canals are filled in the spring in order to determine hydraulic conductivity of an initially dry canal bed (Niswonger et al., 2008). When water first filled the canal, seepage rates were typically one to two orders of magnitude higher than normal, since capillary pressure gradients, caused by the difference in wetness at the sediment surface and deeper in the dry sediment, enhance downward water movement into the streambed sediments. Reaches having relatively higher seepage rates resulted in a relatively slower water velocity of the wetting front. The model developed by Niswonger et al. (2008) used this sensitivity of streambed velocity to seepage rates to determine hydraulic conductivity. Soil hydraulic properties were then transferred to equilibrium conditions, providing a prediction of seepage rates during the summer months. This method was successfully developed and tested by Niswonger in the intermittent Cosumnes River (California), and the Amargosa River (Nevada). Results from our application of this method to three LARV canals in the spring of 2008 look promising, but the final data analysis was not available in time for this report.

Figure 5-5. Variation about the daily mean temperature in top thermocouple during the two weeks before and after canal was observed to be “self-sealed.”
5.2 Influence of Seasonal Changes on Seepage Reduction Estimates

Little is known about the extent to which seepage rates vary naturally throughout the irrigation season. Several factors can affect seepage rates including the concentration and mean particle diameter of suspended sediment, water temperature, water velocity, water stage, groundwater levels, and the use of check structures and other operational procedures employed by canal companies. As a result, the estimates of seepage reduction reported above may be influenced by when the experiments were carried out and by the existing conditions in the canal prior to LA-PAM addition. For example, native seepage rates would be lower following an extended period of sediment-laden water carried by the canal compared to an extended period of clearer, low-sediment water. Therefore, the percent seepage reduction resulting from the addition of LA-PAM would be lower in the former case due to lower native seepage rates prior to application.

In 2007, seepage was repeatedly monitored within three canal reaches to gain a better understanding of how seasonal changes in stage, SSC, and native seepage rates could affect the calculation of seepage reduction. Water depth (stage) was an important driving force, as higher stages exerted greater hydrostatic pressures, increasing seepage. Additionally, higher stages increased the surface area of the canal prism exposed to water, increasing seepage potential. High SSC, estimated using a turbidimeter, indicated the presence of a greater mass of suspended sediment in the water column that can settle to the bottom of the canal over time and reduce the seepage potential by physically blocking pores in the canal bed.

At LSD on the FL (Figure 5-6, Panel A), reduced seepage rates appeared to be primarily related to the lower water levels entering the study reach (LSD100) during the last month of seasonal operation. Water levels at the downstream end (LSD271) of the study reach remained relatively consistent due to the operation of a check structure 15 m downstream of that site. The impact of turbidity on seepage rates could not be determined at this site due to difficulties in obtaining seepage measurements during or near elevated turbidity events.

For the Catlin Canal (CAT) study reach, seepage rates prior to LA-PAM addition in early August 2007 were variable, ranging from less than 0.005 to 0.102 m$^3$ s$^{-1}$ ha$^{-1}$ (Figure 5-6, Panel B). Water level and turbidity were both found to affect seepage rates ($n = 11$):

$$SR = 4.23 \times 10^{-5} \times TU + 0.2001 \times Stg \quad R^2 = 0.77$$

where SR is instantaneous seepage rate (m$^3$ s$^{-1}$ ha$^{-1}$), TU is average daily turbidity in NTU, and Stg is the average daily water stage in meters. The magnitude of stage was more important to seepage rate than turbidity, as stage was highly significant ($p = 0.0065$), whereas turbidity was weakly significant ($p = 0.092$). Operationally, it was difficult to separate the impacts of stage and turbidity on this canal. Water levels in the canal were typically reduced before thunderstorms, providing additional capacity for water entering the canal by overland flow. As a result, thunderstorms that increased turbidity were typically linked with decreased water stage.
Figure 5-6. Instantaneous seepage rate, turbidity, and stage for three canal reaches in the LARV. Dotted line associated with seepage rate was included for visual clarity and does not represent a seepage estimate between the measured points. To place stage on the same axis with turbidity, stage values have been transformed. Panel C is continued on the next page.
Figure 5-6  Instantaneous seepage rate, turbidity, and stage for three canal reaches in the LARV. Dotted line associated with seepage rate was included for visual clarity and does not represent a seepage estimate between the measured points. To place stage on the same axis with turbidity, stage values have been transformed (continued.).
Figure 5-7. Pictures of the RFH banks after (top) the July 30, 2007, high sediment-loading event. For comparative purposes the lower picture shows a downstream section of bank prior to this event.

An example of this is the thunderstorm on July 24, 2007 that increased turbidity in the CAT from 231 NTU to greater than the 1,600 NTU limit of the turbidimeter. Coupled with a stage drop of 0.2 m, seepage was observed to decline by a factor of 10 compared to pre-storm conditions (0.10 to 0.01 m$^3$ s$^{-1}$ ha$^{-1}$). This reduction in seepage was short lived, as seepage returned to 0.051 m$^3$ s$^{-1}$ ha$^{-1}$ on August 2 when turbidity was lower (363 NTU) and stage was higher (+0.22 m). However, the cause of some seepage reductions is not as straightforward, such as the lower seepages observed on July 10 to 11.

For the RFH (Figure 5-6, Panel C), estimated seepage rates varied from 0.005 to 0.050 m$^3$ s$^{-1}$ ha$^{-1}$. Although variability in seepage rates could not be quantitatively explained, they did appear to respond to changes in both turbidity and stage. The first major observed event (Event 1) caused turbidity to increase to 640 NTU on July 20 and 21. High seepage rates were still observed on July 22, but had declined by 26 percent when measured on July 25. This indicated that the levels of sediment mobilized during this moderate event were not enough to make an immediate and sizeable reduction in seepage. The second major turbidity event (Event 2) started with thunderstorms on July 30 and continued beyond August 17. Early thunderstorms during this period impacted the Huerfano and St. Charles sub-watersheds, upstream watersheds that are known to contribute significant quantities of silt.
and clay-sized sediment to the Arkansas River under these conditions. The Huerfano and St. Charles rivers watersheds drain into the river less than 1.6 km (1 mi) and 11 km (7 mi) upstream of the RFH head gate, respectively. The impact that sediment sourced from these upstream watersheds had on the RFH was estimated to be a one-in-three-year event in the Huerfano and a one-in-10-year event in the St. Charles watersheds (Dan Henrichs, Rocky Ford Highline Company, January 2008). Over the first six days after the event, turbidity exceeded the 1,600 NTU maximum reported by the turbidimeters for 70 percent of the time, and did not return to levels below 50 NTU until September 13.

During this event, seepage was severely reduced within the study reach, estimated to be less than 0.005 m$^3$ s$^{-1}$ ha$^{-1}$ on August 6, 15, and 23. Due to the magnitude of this event, the entire length of the RFH canal was affected by high turbidity. However, upper canal reaches closer to the head gate, like the study reach, were more likely to have their seepage impacted. This was because silt-sized sediment carried by the faster river water velocities settled out in the lower water velocities prevalent in the canal. Seepage rates were lowest in August and increased to 0.012 m$^3$ s$^{-1}$ ha$^{-1}$ on September 1. The next two subsequent seepage measurements (September 13 and October 6) were similar despite dropping turbidity levels. However, seepage rates also may have been more affected by a 0.3-m drop in stage that occurred on September 8 and 9.

As discussed above, seepage rates do vary naturally over the irrigation season. Therefore, the simple comparison of seepage rates after LA-PAM application to a single seepage estimate taken before application must be done with caution. The apparent percent seepage reduction could vary greatly depending on antecedent conditions and on the timing of LA-PAM application. Where possible, numerous seepage estimates need to be taken throughout the irrigation season to calculate the actual volume of water lost to seepage, such as presented in the next section. The use of newer technologies, such as temperature as a tracer method, are encouraged because they provide seepage estimates over longer time periods. Once LA-PAM was applied, seepage rates were fairly constant (Figure 5-6) and were not greatly affected by turbidity and water levels for the remainder of the irrigation season. Seepage measurements taken in consecutive years revealed that the seepage benefits from LA-PAM did not typically carry over through the winter when the canals were dewatered.

Seepage rates in untreated canals were sometimes as low as those measured in canal reaches that had been treated with LA-PAM. For example, seepage rates were twice measured to be less than 0.01 m$^3$ s$^{-1}$ ha$^{-1}$ during the 2007 field season. Native seepage rates in CAT can be estimated on a daily basis using the stage/turbidity equation presented above. Assuming a typical turbidity of 225 NTU, results showed that only 13 percent of the 163 operating days between May 23 and November 1, 2007, would have a seepage rate lower than 0.01 m$^3$ s$^{-1}$ ha$^{-1}$ that could not be explained to manually lowered stages in preparation for thunderstorms. For canals such as the RFH, elevated turbidity events did play a role in lowering seepage rates to levels on a par with or below that observed after LA-PAM applications. As discussed above, this somewhat unusual series of thunderstorms greatly reduced seepage rates for approximately 30 days, representing 13 percent of the entire irrigation season or 17 percent of the time that turbidity and stage were monitored (Figure 5-6). Based on these two LARV canals, an early-season application of LA-PAM would be expected to maintain reduced seepage rates for the entire irrigation season whereas
natural processes might result in reduced seepage rates for up to 13 to 17 percent of the irrigation season.

5.3 Estimated Actual and Potential Seepage Volume Reduction

Three approaches were used to estimate seepage reduction volumes resulting from the addition of LA-PAM to several canals in the LARV. First, seepage reduction in CAT-2 was estimated for the treated section for the month following LA-PAM application based on 2007 field data. Second, annual seepage reduction within three studied canal reaches was estimated using the volume balance procedure based on 2007 field measurements. Third, seepage reduction along the entire length of several canals in the LARV was estimated based on computer modeling.

5.3.1 Estimated Seepage Volume Reduction in Study Reaches

In 2007, seepage measurements were made on LARV canal reaches that were expected to have high seepage rates without application of LA-PAM. Seepage measurements were collected throughout the water year to incorporate natural seepage patterns that characterize each studied canal reach. Continuous measurements were not possible, so seepage rates were interpolated to estimate seepage volume. For the Catlin Canal, seepage volumes were estimated based on the numerous discharge measurements that were conducted one month prior to and one month after LA-PAM application (CAT-2) (Table 5-4). The application of LA-PAM resulted in a 70-percent seepage reduction and a water savings of 62,015 m$^3$ km$^{-1}$ for the first month after application.

<table>
<thead>
<tr>
<th>No. Measurements</th>
<th>Estimated Total Seepage Volume (per month)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Post-PAM</td>
<td>342,543</td>
</tr>
<tr>
<td>Pre-PAM</td>
<td>104,408</td>
</tr>
<tr>
<td>Water Savings (per month)</td>
<td>238,135</td>
</tr>
</tbody>
</table>

Seepage volume reduction was also estimated for one reach of the RFH and two reaches of the FL based on field data collected during 2007. Linear anionic PAM was not applied to these canal reaches in 2007; however, LA-PAM was applied to the RFH in 2006 (RFH-2). Seepage measurements show that the seepage reduction benefit observed in 2006 at RFH did not carry over into 2007. Seepage reduction volumes are estimated at 35 and 80 percent levels of seepage reduction (Table 5-5), typical values observed during the 2006 and 2007 field studies.

5.3.2 Annual Seepage Volume Reduction Estimated for LARV Canals

An estimate of the average total annual water volume diverted by LARV canals from the Arkansas River was calculated from hourly flow rate measurements posted online at the Colorado Division of Water Resources website (http://www.dwr.state.co.us/). Diversion flow rates over the past 25 years were used to calculate average annual total diverted volumes into LARV canals from the Arkansas River (Table 5-6). Based upon calibrated finite-difference computer models of the LARV created by CSU, it was estimated that on average 20 percent to 30 percent of the total diverted flow rate from the river seeps from the wetted perimeter of
LARV canals (Gates et al., 2002, 2006; Burkhalter and Gates 2005, 2006). Groundwater table elevation data, canal water surface elevation data, and hydraulic conductivity data of the soil surrounding the canals were used to model the LARV and estimate the seepage percentages. Based on estimated seepage reductions of 35 and 80 percent observed in the field, the models were used to predict annual seepage reduction volume from LA-PAM application to the entire length of the major LARV canals (Table 5-6).
Table 5-5. Estimated volume of annual seepage reduction from LARV canals with LA-PAM application. Estimates are based on the specific canal reach under study.

<table>
<thead>
<tr>
<th>Study Reach</th>
<th>Number of Measurements</th>
<th>Estimated Annual Seepage Volume (10^3 m³)</th>
<th>Estimated Annual Seepage Volume (10^3 m³ km⁻¹)</th>
<th>Estimated Annual Seepage Reduction Volume (10^3 m³)</th>
<th>Estimated Annual Seepage Reduction Volume (10^3 m³ km⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>RFH</td>
<td>15</td>
<td>4,487</td>
<td>427</td>
<td>1,570</td>
<td>150</td>
</tr>
<tr>
<td>FL Las Animas Division</td>
<td>2</td>
<td>18,399</td>
<td>460</td>
<td>6,440</td>
<td>161</td>
</tr>
<tr>
<td>FL Limestone Division</td>
<td>4</td>
<td>24,365</td>
<td>886</td>
<td>8,528</td>
<td>310</td>
</tr>
</tbody>
</table>

Table 5-6. Estimated volume of annual seepage reduction from LARV canals with LA-PAM application to the entire canal. Seepage was measured by differential surface water measurements conducted by ADV up through 2005 and by ADCP in 2006 and 2007.

<table>
<thead>
<tr>
<th>Canal</th>
<th>Average Annual Volume Diverted (10⁶ m³)</th>
<th>Estimated Annual Volume of Seepage (10⁶ m³)</th>
<th>Estimated Volume of Annual Seepage Reduction Using PAM (10⁶ m³)</th>
<th>35% Seepage Reduction</th>
<th>85% Seepage Reduction</th>
</tr>
</thead>
<tbody>
<tr>
<td>Amity</td>
<td>101.2</td>
<td>20.2</td>
<td>7.1</td>
<td>10.6</td>
<td>17.2</td>
</tr>
<tr>
<td>Bessemer</td>
<td>72.0</td>
<td>14.4</td>
<td>5.0</td>
<td>7.6</td>
<td>12.2</td>
</tr>
<tr>
<td>Buffalo</td>
<td>23.5</td>
<td>4.7</td>
<td>1.6</td>
<td>2.5</td>
<td>4.0</td>
</tr>
<tr>
<td>Catlin</td>
<td>103.3</td>
<td>20.7</td>
<td>7.2</td>
<td>10.9</td>
<td>17.6</td>
</tr>
<tr>
<td>Colorado</td>
<td>109.9</td>
<td>22.0</td>
<td>7.7</td>
<td>11.5</td>
<td>18.7</td>
</tr>
<tr>
<td>Fort Bent</td>
<td>20.2</td>
<td>4.0</td>
<td>1.4</td>
<td>2.1</td>
<td>3.4</td>
</tr>
<tr>
<td>Fort Lyon</td>
<td>340.2</td>
<td>68.0</td>
<td>23.8</td>
<td>35.7</td>
<td>57.8</td>
</tr>
<tr>
<td>Lamar</td>
<td>48.3</td>
<td>9.7</td>
<td>3.4</td>
<td>5.1</td>
<td>8.2</td>
</tr>
<tr>
<td>Las Animas Consol.</td>
<td>31.8</td>
<td>6.4</td>
<td>2.2</td>
<td>3.3</td>
<td>5.4</td>
</tr>
<tr>
<td>Manvel</td>
<td>2.2</td>
<td>0.4</td>
<td>0.2</td>
<td>0.2</td>
<td>0.4</td>
</tr>
<tr>
<td>Otero</td>
<td>11.6</td>
<td>2.3</td>
<td>0.8</td>
<td>1.2</td>
<td>2.0</td>
</tr>
<tr>
<td>Oxford Farmers</td>
<td>30.0</td>
<td>6.0</td>
<td>2.1</td>
<td>3.2</td>
<td>5.1</td>
</tr>
<tr>
<td>Rocky Ford</td>
<td>53.4</td>
<td>10.7</td>
<td>3.7</td>
<td>5.6</td>
<td>9.1</td>
</tr>
<tr>
<td>Rocky Ford Highline</td>
<td>95.0</td>
<td>19.0</td>
<td>6.7</td>
<td>10.0</td>
<td>16.2</td>
</tr>
<tr>
<td><strong>Average</strong></td>
<td><strong>74.5</strong></td>
<td><strong>14.9</strong></td>
<td><strong>5.2</strong></td>
<td><strong>7.8</strong></td>
<td><strong>12.7</strong></td>
</tr>
</tbody>
</table>
The total annual cost of applying LA-PAM to water delivery irrigation canals includes LA-PAM in granular form, labor, necessary equipment, and vehicle operation. The following section presents estimates of maximum and minimum prices of LA-PAM application based upon average application time requirements from the 2006 and 2007 field studies in the LARV of Colorado (J. Swihart, U.S. Bureau of Reclamation, personal communication, 2007). Costs are calculated in 2007 US dollars. The analysis below shows the cost of LA-PAM application to range from $78 km\(^{-1}\) for smaller canals to $213 km\(^{-1}\) for larger canals depending on the application rate and methodology used. Also, annual cost of LA-PAM is estimated to be less than 3 percent of the total annualized cost of traditional methods such as concrete and geomembranes.

### 6.1 LA-PAM, Labor, and Equipment

Although the price of granular LA-PAM varies, the typical cost was roughly $8.82 kg\(^{-1}\) ($4.00 lb\(^{-1}\)) (J. Swihart, U.S. Bureau of Reclamation, personal communication, 2007). Two types of spreaders were used: hand-held or automatic. Hand-held spreaders can be purchased for about $10 each, chest-mounted spreaders for $150, and an automatic spreader can be purchased for about $400. An automatic spreader must be mounted on the front of the boat, which would cost approximately $300 for mounting supplies and fabrication. A boat was preferred for the application process for several reasons. Application was much more rapid, reducing labor costs, LA-PAM was dispersed more evenly and continuously on the water surface, and the turbulence created by the outboard motor more evenly mixed and distributed the LA-PAM into the water. A truck was necessary to act as a pace-setter and to supply bags of LA-PAM. The rate of LA-PAM application was controlled by adding a specified weight of LA-PAM to the spreader(s) over the distance between truck stops. Vehicle efficiency was approximately 4.25 km L\(^{-1}\) (10 mi gal\(^{-1}\)) or less. Boat fuel economy was approximately 0.85 km L\(^{-1}\) (2 mi gal\(^{-1}\)) for a 10-horsepower outboard motor. The average cost of gasoline was assumed to be $0.86 L\(^{-1}\) ($3.25 gal\(^{-1}\)).

The cost of labor depended on several factors. For small canals, two technicians applied the polymer. One technician applied the LA-PAM by walking in or adjacent to the canal while the other drove the truck and resupplied the polymer applicator. As the canal size grew, two technicians were used to walk the canal side-by-side to improve the applicator coverage. In larger canals where a motorized boat was used, four technicians were required to complete the application process: one to two technicians to spread the LA-PAM over the canal water from inside a boat; one to drive the boat; and another person to drive the supply truck alongside the canal road.

Labor costs also depended on the rate at which LA-PAM was applied to the canal. The slowest application approach was walking the canal either within the canal prism (0.5 km hr\(^{-1}\)) or along the banks of the canal (0.6 to 1.3 km hr\(^{-1}\)). With handheld spreaders in a motorized boat, two workers were required to apply the polymer at a typical rate of 1.6 canal km hr\(^{-1}\) (1 canal mi hr\(^{-1}\)). Using a boat-mounted automatic spreader, only one person was required to apply the polymer at a typical rate of 4.8 canal km hr\(^{-1}\) (3 canal mi hr\(^{-1}\)). However, the actual speed of motorized-assisted applications was dependant upon the number of obstacles (fences, bridges, etc.) in the canal, the experience of the technicians,
dependability of the equipment, and size of the outboard motor. Applications during these studies varied from 0.14 hr km\(^{-1}\) (0.22 hr mi\(^{-1}\)) to 0.76 hr km\(^{-1}\) (1.22 hr mi\(^{-1}\)) and averaged 0.47 hr km\(^{-1}\) (0.75 hr mi\(^{-1}\)). To convert the associated labor costs, the assumption was made that an average application would cover 16 km (10 mi). Two technicians could complete the tasks of application planning, mobilization, and equipment cleaning and maintenance over a total time period of five hours. If all four workers make between $15.00 and $20.00 hr\(^{-1}\), the total cost of labor would average $70.00 hr\(^{-1}\).

The technicians applying the polymer wore full protective gear while the LA-PAM application took place in accordance with the application guidelines (Susfalk et al., 2007). When a boat was used, all persons in the boat wore protective gear. Eye-protective goggles and paper respiratory masks cost about $10 per person per application. Body protective suits such as Tyvek coveralls ($55 for a case of 25) should be worn and all areas of the skin should be covered. A head cover is recommended to protect the face and head from contact with LA-PAM. It should be noted that wind, either natural or that due to boat movement, will cause the dispersal of LA-PAM on nearby surfaces. For example, granules of LA-PAM built up on the floor of the boat causing the floor to become extremely slippery.

6.2 Total Cost Estimate

The total cost of LA-PAM application varies based on the canal width, wetted perimeter, and application technique. The wetted perimeter is factored into the application rate and, consequently, the price of LA-PAM application to that canal. Using a boat-based application method, the least expensive LA-PAM application would be a rate of 11.2 kg ha\(^{-1}\) (10 lbs ac\(^{-1}\)) using an automatic spreader. The most expensive LA-PAM application would be a rate of 16.8 kg ha\(^{-1}\) (15 lbs ac\(^{-1}\)) using manual spreaders. The use of manual spreaders on the boat resulted in slower application rates than in applications using automatic spreaders, and it also required the presence of a second technician. As a result, labor costs are significantly greater when using hand-powered spreaders on a boat.

A wide canal is less expensive than a narrow canal under the assumption that a greater area of canal is treated for the same cost as a narrow canal, effectively reducing labor, gasoline, and vehicle costs per wetted perimeter area (Table 6-1). The vehicle cost is the same for both techniques because it is simply a rate of vehicle usage (cost per mile), which is not a function of application speed.
Table 6-1. Summary of estimated LA-PAM application cost at five sites.

<table>
<thead>
<tr>
<th></th>
<th>CAT</th>
<th>LAM</th>
<th>RFH</th>
<th>SD</th>
<th>KC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Typical $Q$ (m$^3$ s$^{-1}$)</td>
<td>3.1</td>
<td>1.4</td>
<td>4.1</td>
<td>0.10</td>
<td>0.15</td>
</tr>
<tr>
<td>Wetted Perimeter (m)</td>
<td>7.6</td>
<td>6.1</td>
<td>10.7</td>
<td>3.1</td>
<td>2.5</td>
</tr>
<tr>
<td>Unit Area (ha km$^{-1}$)</td>
<td>0.77</td>
<td>0.61</td>
<td>1.07</td>
<td>0.30</td>
<td>0.25</td>
</tr>
<tr>
<td>LA-PAM Application Rate ($km$^{-1}$)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>11.2 kg ha$^{-1}$ at $8.82$ kg$^{-1}$</td>
<td>$75.76$</td>
<td>$60.61$</td>
<td>$106.06$</td>
<td>$30.11$</td>
<td>$24.70$</td>
</tr>
<tr>
<td>16.8 kg ha$^{-1}$ at $8.82$ kg$^{-1}$</td>
<td>$113.64$</td>
<td>$90.91$</td>
<td>$159.09$</td>
<td>$45.16$</td>
<td>$37.05$</td>
</tr>
<tr>
<td>Labor ($km$^{-1}$)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Auto Spreader</td>
<td>$30.32$</td>
<td>$30.32$</td>
<td>$30.32$</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
<td>Manual Spreader</td>
<td>$52.98$</td>
<td>$52.98$</td>
<td>$52.98$</td>
<td>$52.98$</td>
<td>$52.98$</td>
</tr>
<tr>
<td>Vehicle Use ($km$^{-1}$)</td>
<td>$0.31$</td>
<td>$0.31$</td>
<td>$0.31$</td>
<td>$0.31$</td>
<td>$0.31$</td>
</tr>
<tr>
<td>Gasoline ($km$^{-1}$)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Truck</td>
<td>$0.20$</td>
<td>$0.20$</td>
<td>$0.20$</td>
<td>$0.20$</td>
<td>$0.20$</td>
</tr>
<tr>
<td>Boat</td>
<td>$1.02$</td>
<td>$1.02$</td>
<td>$1.02$</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
<td>Total Cost ($km$^{-1}$)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Minimum</td>
<td>$107.61$</td>
<td>$92.46$</td>
<td>$137.91$</td>
<td>$83.60$</td>
<td>$78.19$</td>
</tr>
<tr>
<td>Maximum</td>
<td>$168.15$</td>
<td>$145.42$</td>
<td>$213.60$</td>
<td>$98.65$</td>
<td>$90.53$</td>
</tr>
<tr>
<td>Average</td>
<td>$137.88$</td>
<td>$118.94$</td>
<td>$175.76$</td>
<td>$91.13$</td>
<td>$84.36$</td>
</tr>
</tbody>
</table>

6.3 Comparison with Conventional Canal Linings

Conventional canal lining materials include concrete, fluid-applied membrane, and geomembranes. These materials have a much greater cost per area for construction and maintenance than LA-PAM applications; however, LA-PAM must be applied at least once per year, while the alternative materials will last between 10 and 60 years (Swihart and Haynes, 2002). Nevertheless, the total annualized cost of LA-PAM application (assuming one application per water year) is only 0.14 to 3.48 percent the total annualized cost of conventional lining materials. For LA-PAM applications the greatest annual cost is $202 ha$^{-1}$ ($82 ac$^{-1}$) (Table 6-2). This cost is 0.3, 3.5, 2.2, and 2.9 percent of the total annualized cost of fluid-applied membrane, concrete, exposed geomembrane, and geomembrane with a concrete cover canal lining, respectively.

The total annualized cost for LA-PAM applications (Table 6-2) includes granular LA-PAM, labor, gasoline, application equipment (boat and spreaders), vehicle expenses, and protective gear. The equipment prices include the cost of a new boat and spreader that are assumed to function properly for 10 years at a work rate of 101 ha yr$^{-1}$ (250 ac yr$^{-1}$). It was assumed that one 101 ha (250 ac) is approximately equal to 129 km (80 mi) of canal with an average wetted perimeter of 7.6 m (25 ft) or 80.5 km (50 miles) of canal with an average wetted perimeter of 12.2 m (40 ft). Linear anionic PAM application experiments showed that protective gear remains in usable condition for roughly 16 km (10 mi) of application or one full day, after which time new safety gear is needed.
Table 6-2. Comparison of LA-PAM and conventional canal linings. All lining estimates, except that of LA-PAM, were provided by Swihart and Haynes (2002). These prices, adjusted to the 2007 United States dollar, using the United States Department of Labor inflation rates (USDL, 2007).

<table>
<thead>
<tr>
<th>Lining Material</th>
<th>Annualized Construction Cost ($ ha⁻¹ yr⁻¹)</th>
<th>Annual Maintenance Cost ($ ha⁻¹ yr⁻¹)</th>
<th>Durability (Yrs)</th>
<th>Effective Seepage Reduction</th>
<th>Total Annualized Cost ($ ha⁻¹ yr⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fluid-applied membrane</td>
<td>$80,629</td>
<td>$1,248</td>
<td>10 - 15 yrs</td>
<td>90%</td>
<td>$81,877</td>
</tr>
<tr>
<td>Concrete</td>
<td>$5,178</td>
<td>$624</td>
<td>40 - 60 yrs</td>
<td>70%</td>
<td>$5,802</td>
</tr>
<tr>
<td>Exposed Geomembrane</td>
<td>$8,110</td>
<td>$1,248</td>
<td>10 - 25 yrs</td>
<td>90%</td>
<td>$9,357</td>
</tr>
<tr>
<td>Geomembrane with Concrete Cover</td>
<td>$6,238</td>
<td>$624</td>
<td>40 - 60 yrs</td>
<td>95%</td>
<td>$6,862</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Re-Application Rate (appl yr⁻¹)</th>
<th>Durability (Yrs)</th>
<th>Effective Seepage Reduction*</th>
<th>Total Annualized Cost per Application ($ ha⁻¹ yr⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Polyacrylamide (LA-PAM)</td>
<td>1 - 2 times</td>
<td>&lt; 1 year</td>
<td>$111 - $202</td>
</tr>
</tbody>
</table>

* For one PAM application per water year in the 2006 and 2007 studies
7.0 POTENTIAL RISKS OF LA-PAM APPLICATION I: LA-PAM AND AMD RELEASE

The application of LA-PAM into unlined water delivery canals warrants consideration of potential environmental and health risks. The actual risk of LA-PAM use must be determined on a site-by-site basis, because factors affecting LA-PAM efficacy are site-specific. The use of proper application techniques (Susfalk et al., 2007) and an understanding of the interaction between application techniques and environmental conditions introduced in Chapter 4 and discussed in detail in Chapter 9 can be used to manage risk.

The most likely human health risk associated with the LA-PAM polymer occurs during the actual application process when workers come directly into contact with the polymer. The use of proper safety precautions (Susfalk et al., 2007) minimizes this risk. After application, dissolved LA-PAM can be removed from the water column by either reacting with and flocculating suspended sediments or by adsorbing to the surface sediment along the canal prism (Malik et al., 1991). Therefore, the only immediate routes for LA-PAM to leave the treated canal prism are: downstream transport of the hydrated polymer beyond the lower application boundary, transport through open offtake gates that could move the polymer into on-farm canals and agricultural fields, and use of the water by livestock. Section 7.1 details the concentrations of LA-PAM that were observed to remain in the water column below the application reach during several application experiments.

The most important perceived health risk associated with the use of LA-PAM is the release of the residual acrylamide monomer (AMD) into canal water and subsequently into groundwater. Several investigations were conducted to better understand the release and transport of AMD. First, a discussion of AMD concentrations observed in the water column is presented in Section 7.2.1. Second, the results from a field-scale tracer study were used to assess the likelihood of AMD entering adjacent groundwater wells in Section 7.2.2. Third, laboratory and model studies were coupled in Section 7.2.3 to better understand the transport and degradation of AMD in groundwater. Finally, a summary of the results from a risk-characterization study regarding LA-PAM addition to unlined canals is presented in Section 7.2.4.

The results from the studies presented below show that: 1) elevated LA-PAM concentrations were not always observed; however, when higher concentrations were observed, LA-PAM concentrations were more likely to be greater in smaller canals (16 mg L\(^{-1}\) maximum) compared to larger canals (4.8 mg L\(^{-1}\) maximum); 2) dissolved LA-PAM that remains in the water column could travel a significant distance downstream; 3) low AMD concentrations observed in canals were orders of magnitude below the chronic levels needed to impact human health; 4) the transfer of AMD from the canal to groundwater decreased with the formation of PAM-sediment flocs and the reduction in seepage rates; 5) AMD concentrations in groundwater were diluted by over four orders of magnitude by groundwater and canal water, as assessed by an AMD-surrogate under field conditions; 6) bacterial degradation was a primary factor, resulting in an AMD half-life in natural systems of 30 to 42 hours as determined by laboratory studies; and 7) transport models indicated that an AMD concentration 10 times greater than the maximum concentration observed in canal water would be undetectable within 25 m of the canal due to microbial
degradation and dilution processes. Therefore, the contamination of groundwater by AMD associated with the application of LA-PAM to water delivery canals using the methods of Susfalk et al. (2007) was considered to be very unlikely.

7.1 Concentration of LA-PAM in Canal Waters

Between 2005 and 2007, LA-PAM was applied to 17 canal reaches. Specific application procedures and canal abbreviations are found in Table 2-2. During these applications, 604 water samples were collected to assess the total concentration of LA-PAM in canal water. Loss of polymer to the environment was managed by shutting down offtake gates in and downstream of the application zone and by applying only the amount of LA-PAM needed to achieve seepage reduction (Susfalk et al. 2007). The target application rate for most applications was 11 kg ha⁻¹ (10 lbs ca⁻¹) of LA-PAM based on the wetted perimeter. This rate was chosen because laboratory (Young et al., 2007b; Susfalk et al., 2005), furrow-scale experiments, and previous applications (Max Schmidt, Water Solutions of Colorado, personal communication, 2005) indicate it to be effective at reducing seepage rates, though Natural Resources Conservation Service (NRCS) guidelines (NRCS, 2005) permit the use of up to 44 kg ha⁻¹ (40 lbs ca⁻¹) of LA-PAM. The lower rate was preferred because less LA-PAM would be released into the environment.

Aggregated results from these field applications showed several important trends. First, two-thirds of the 574 noncontrol samples collected for LA-PAM analysis were less than the U.S. EPA limit of 1.0 mg L⁻¹, and 287 of the 574 samples were below the 0.1 mg L⁻¹ detection limit. Concentrations of LA-PAM remained below detection limits when sediment concentrations were sufficient to react with the entire mass of added polymer. This was shown at two canals that had differing suspended sediment concentrations during polymer application to either the same canal reach two consecutive years (CAT) or application to two adjacent canal reaches of the same canal within days of each other (HID). The use of suspended sediment augmentation to increase suspended sediment concentrations in the water column, however, was not always sufficient to promote the PAM-sediment floc formation that leads to reduced LA-PAM concentrations (see Chapter 9).

When measurable, LA-PAM concentrations decreased over time if suspended sediment levels were sufficient to form PAM-sediment flocs, were more likely to be elevated in smaller canals (16 mg L⁻¹ maximum) compared to larger canals (4.8 mg L⁻¹ maximum), and were only detectable at the bottom of the application zone for up to six hours after the cessation of polymer addition, though this value varied depending on the total time it took to apply LA-PAM and the water velocity in the canal. In both small and large canals, dissolved LA-PAM that remains in the water column can travel substantial distances downstream, as observed at MD (2.8 km) and HID-2 (3.6 km).

The detailed methods used for LA-PAM application and water quality sampling were presented in Section 3.4, with site-specific modifications to the methods noted in the individual site descriptions in Section 2.3. Results are discussed below beginning with aggregated LA-PAM concentrations followed by results from the individual experiments carried out in smaller and larger volume canals.
7.1.1. Aggregated Results

During the field application experiments, 574 samples were collected for LA-PAM analysis. These samples were not collected in control reaches. Of these samples, 66 percent were found to have LA-PAM concentrations below the U.S. EPA limit of 1.0 mg L\(^{-1}\), and 50 percent were below the 0.1 mg L\(^{-1}\) detection limit (Figure 7-1). The sampling protocol dictated a greater proportion of samples to be collected during the first few hours of LA-PAM application when concentrations were most likely to be highest. The majority of these samples were collected just below the downstream boundary of the LA-PAM application zone, so that the concentration of polymer leaving the treatment zone could be monitored.

On average, concentration of LA-PAM was initially high and then decreased over time. Figure 7-2 shows that LA-PAM, on average, was higher than the EPA limit of 1 mg L\(^{-1}\) for only the first six hours and was below the 0.1 mg L\(^{-1}\) detection limit after 18 hours. “Average” concentrations can be misleading, however, because site-by-site LA-PAM concentrations were found to be highly variable. Figure 7-3 presents data that have been normalized by site; each polymer application contributes a single average number for a given elapsed time period. The colors within each bar represent the average concentration at a site during a given time period. The total height of each bar represents the number of observations, and decreases with time.

![Samples Grouped by LA-PAM Concentration](image)

Figure 7-1. LA-PAM concentrations measured during all polymer applications grouped by concentration. The majority of the samples were collected at the most downstream boundary of LA-PAM application, however, this aggregated dataset does include samples collected from locations further downstream.
Figure 7-2. Average LA-PAM concentration measured just below the downstream boundary of the LA-PAM application zone.

Figure 7-3. Average LA-PAM normalized by experiment. Each polymer application experiment contributes a single average LA-PAM value for a given elapsed time period.
Several conclusions can be drawn from Figure 7-3. First, highly elevated concentrations of LA-PAM (> 4 mg L\(^{-1}\)) were only observed at one or two sites during the first six hours after application. After six hours, the average concentration did not exceed 4 mg L\(^{-1}\). Second, the majority of sites had LA-PAM concentrations that were not detectable (light blue in Figure 7-3) or ranged from 0.1 to 1.0 mg L\(^{-1}\) (red) during the first hour. This large variability indicates the role that differing environmental and physical factors, such as turbidity, water chemistry, flow rate, and wetted perimeter have on LA-PAM hydration and flocculation. Results from several of the LA-PAM application studies were divided in low and high volume canals.

7.1.2. Field Case Studies: Low-volume Canals

Five LA-PAM application experiments were conducted in canals with flow below 2.1 m\(^3\) s\(^{-1}\) (75 ft\(^3\) s\(^{-1}\)): SD, KC-1, MD, LAM-1, and LAM-2 (Figures 7-4, 7-5, 7-6, 7-7, and 7-8, respectively). Sample locations are described in Section 2.3. Samples with PAM concentrations below the detection limit are plotted just above the x-axis in the following figures.

The maximum LA-PAM concentration observed at SD was 3.0 mg L\(^{-1}\) at 500 m downstream, five hours after polymer application finished (Figure 7-4). This peak concentration was observed at sites both upstream and downstream. Given the average water velocity of 0.07 m s\(^{-1}\) in the treatment section, this peak LA-PAM concentration originated 1,197 m upstream, near the upper polymer addition boundary. During the first several application attempts, the polymer application technicians found that they were under-applying the polymer as they approached the end of the treatment reach. They would then apply a greater rate of LA-PAM to “make up” for the under-application, resulting in the higher concentrations in LA-PAM observed later downstream. Linear anionic PAM concentrations fell below the detection limit by 10 hours at 0 m, and were not measurable in the observation reach 16 hours after LA-PAM was initially applied.

At KC-1, the highest LA-PAM concentrations of all sites studied were observed, peaking at 16 mg L\(^{-1}\) and remaining elevated above 1 mg L\(^{-1}\) for five hours (Figure 7-5). Several factors may have contributed to the high polymer concentrations. First, cold water temperature (21° C) slowed the rate of LA-PAM hydration. Second, flocs that formed may have remained suspended in the water column or saltated along the bottom of the canal given the turbulent flow and relatively fast water velocities (0.33 m s\(^{-1}\)) for such a small canal (2 m wetted perimeter). Compared with SD, KC-1 carried 33 percent more flow, with 35 percent less wetted perimeter, and had water velocities that were five times greater. Finally, PAM-sediment flocs may not have readily formed due to low suspended sediment levels that were naturally dropping during the application. At the control site (not exposed to the polymer), suspended sediment concentrations dropped from 96 mg L\(^{-1}\) near the start of the application to 24 mg L\(^{-1}\) at 3.6 elapsed hours. However, the suspended sediment concentration was locally elevated by using a backhoe to mix up canal bottom sediments from 30 to 170 m upstream of the active LA-PAM application point. Although not measured directly in this early experiment, elevated suspended sediment concentrations were likely greater than 750 mg L\(^{-1}\) for a short time (this value was observed at other sites where sediment dredging was used). The attempt at creating localized elevated sediment levels appeared unsuccessful, given the high concentration of both LA-PAM and suspended
sediment, which should have been lowered by the formation of PAM-flocs. The lack of floc formation may be explained by water temperature delaying the hydration of LA-PAM.

Figure 7-4. Plot of LA-PAM concentrations, elapsed time, and sample location during LA-PAM application at SD on August 25, 2005. Application began at 10:00 am and lasted for just over 3 hours. Samples were collected at 0, 100, 250, 500, 940, and 1479 m downstream of the application boundary. For a detailed description of the determination of sampling frequency see Section 2.3.8.

Figure 7-5. Plot of LA-PAM concentrations, elapsed time, and sample location during LA-PAM application at KC-1 on July 18, 2005. Application began at 10:30 a.m. and lasted for 2 hours. Samples were collected at 30, 120, and 805 m downstream of the application boundary.
Figure 7-6. Plot of LA-PAM concentrations, elapsed time, and sample location during LA-PAM application at MD on June 23, 2005. Application began at 11:00 a.m. and lasted for 2.5 hours. Samples were collected at 30, 150, 805, and 2,816 m downstream of the application boundary.

Figure 7-7. Plot of PAM concentration and elapsed time at LAM-1 during LA-PAM application on May 18, 2006. Application lasted for 8.3 hours. Samples were collected at a single location downstream of the application boundary.
Figure 7-8. Plot of PAM concentration and elapsed time at LAM-2 during the second LA-PAM application on June 7, 2006. Application lasted for 9 hours. Samples were collected at a single location downstream of the application boundary.

Similar to the other canals studied, the concentration of LA-PAM at MD (Figure 7-6) was also elevated. The maximum LA-PAM concentration observed was 8.6 mg L\(^{-1}\), measured 30 m downstream at 2.9 hours after the application began. By five hours, LA-PAM concentrations had returned to below detection limits at the three upstream stations (30 to 805 m). As with the KC-1 experiment, water temperature was low (20 °C) with very high velocities (0.91 m s\(^{-1}\)), promoting the downstream transport of LA-PAM.

Two LA-PAM application experiments were conducted at LAM (LAM-1 and LAM-2; Table 7-1). Sediment addition by raking increased suspended sediment from an estimated 67 mg L\(^{-1}\) to 2,900 mg L\(^{-1}\), as determined by several depth-width integrated samples taken throughout the experiment. Although raking was effective in creating an instantaneous suspended sediment peak, visual observation showed that the suspended sediment distribution across the canal was highly variable and that the bulk of the added sediment had dropped out less than 20 m downstream. The elevated LA-PAM concentrations observed for the duration of this experiment (Figure 7-7) indicated that this sediment addition approach was not very effective at promoting the formation of PAM-sediment flocs. During the second application at LAM-2, suspended sediment was not augmented, as concentrations averaged 768 plus or minus 39 mg L\(^{-1}\) (n = 5) at the control site. Linear anionic PAM concentrations initially peaked at nearly 4 mg L\(^{-1}\) and consistently declined over time (Figure 7-8). This decline in polymer concentration over time at LAM-2 was indicative of the removal of LA-PAM from the water column through PAM-sediment floc formation, which did not occur at LAM-1.
Table 7-1. Background properties of the canal reaches prior to LA-PAM application.

<table>
<thead>
<tr>
<th>Canal Reach</th>
<th>Length (km)</th>
<th>Flow Rate (m$^3$s$^{-1}$)</th>
<th>Background Suspended Sediment Concentration (mg L$^{-1}$)</th>
<th>Wetted Perimeter (m)</th>
<th>LA-PAM Application Rate (kg ha$^{-1}$) (lbs acre$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low-volume Canals</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>SD</td>
<td>1.4</td>
<td>0.1</td>
<td>16</td>
<td>3.0</td>
<td>12.4 (11.1)</td>
</tr>
<tr>
<td>KC-1</td>
<td>2.4</td>
<td>0.2</td>
<td>24 to 96</td>
<td>2.5</td>
<td>11.0 (10.0)</td>
</tr>
<tr>
<td>MD</td>
<td>3.2</td>
<td>0.8</td>
<td>163</td>
<td>2.6</td>
<td>11.0 (9.8)</td>
</tr>
<tr>
<td>LAM-1</td>
<td>4.5</td>
<td>0.6</td>
<td>&lt;60*</td>
<td>7.7</td>
<td>11.0 (10.0)</td>
</tr>
<tr>
<td>LAM-2</td>
<td>9.3</td>
<td>1.4</td>
<td>768</td>
<td>6.5</td>
<td>12.6 (11.2)</td>
</tr>
<tr>
<td>High-volume Canals</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CAT-1</td>
<td>4.4</td>
<td>3.1</td>
<td>471</td>
<td>6.4</td>
<td>18.3 (16.3)</td>
</tr>
<tr>
<td>CAT-2</td>
<td>4.4</td>
<td>3.3</td>
<td>1,745</td>
<td>6.7</td>
<td>12.1 (10.8)</td>
</tr>
<tr>
<td>RFH-1</td>
<td>16.7</td>
<td>3.1</td>
<td>144</td>
<td>10.7</td>
<td>13.3 (11.9)</td>
</tr>
<tr>
<td>RFH-2</td>
<td>4.0</td>
<td>3.3</td>
<td>570</td>
<td>12.3</td>
<td>14.3 (12.7)</td>
</tr>
<tr>
<td>UNC</td>
<td>6.3</td>
<td>21.8</td>
<td>27</td>
<td>29.6</td>
<td>9.2 (8.2)</td>
</tr>
<tr>
<td>HID-1†</td>
<td>4.5</td>
<td>16.9</td>
<td>78</td>
<td>15.8</td>
<td>13.9 (12.4)</td>
</tr>
<tr>
<td>HID-2†</td>
<td>4.3</td>
<td>17.6</td>
<td>31</td>
<td>17.1</td>
<td>13.6 (12.1)</td>
</tr>
</tbody>
</table>

*Estimated using turbidity values using data from LAM-2
† Stationary application method was used

7.1.3. Field Case Studies: High-volume Canals

Liner anionic PAM was also applied to seven larger-volume canals with flow ranging from 3.1 to 21.8 m$^3$s$^{-1}$: GVIC, UNC, RFH-1, HID-1, HID-2, CAT-1, and CAT-2. The HID experiments were conducted near Billings, MT, the GVIC and UNC in western Colorado, and the remaining experiments in the LARV in eastern Colorado.

At the RFH (RFH; Table 7-1), only one water sample, collected after one hour elapsed time at 0.5 km downstream, had a LA-PAM concentration (2.4 mg L$^{-1}$) greater than the U.S. EPA 1 mg L$^{-1}$ limit. A second application at the upper reaches of this canal was conducted on July 20, 2006. However, no water chemistry samples were collected.

Two experiments were conducted along the same reach of CAT (CAT; Table 7-1). The greatest concentration of LA-PAM was 2.1 mg L$^{-1}$, 60 min after LA-PAM application had started. Concentrations remained about 1 mg L$^{-1}$ for up to five hours (Figure 7-10). A second application was conducted on August 7, 2007, using the same methodology. All water samples collected during the 2007 study remained below the detection limits of LA-PAM. The greatest difference between the two experiments was that CAT-2 had nearly four times more SSC than CAT-1 (Table 7-1), so the additional sediment during CAT-2 promoted the formation of PAM-sediment flocs, effectively removing LA-PAM from the water column. Suspended sediment concentrations were high enough to be only slightly depressed after polymer addition. A clear zone never fully developed.

Two experiments were also conducted on adjacent reaches of the HID. At HID-1, samples collected from three sites (0.8 to 7.4 km below the application boundary) remained below the detection limit for LA-PAM. At HID-2, LA-PAM was measurable, but below the U.S. EPA limit for approximately two hours in samples collected at 1.9 and 3.6 km downstream (Figure 7-11).
Similar to the CAT studies, these results highlight the role that different suspended sediment concentrations have on LA-PAM concentrations. Suspended sediment at HID-1 was high enough to react with the full mass of LA-PAM added to the canal, thereby reducing polymer concentrations to undetectable levels. At HID-2, however, the suspended sediment concentrations were lower and not capable of reacting with all the LA-PAM added, resulting in measurable polymer concentrations. Based on the experiences at HID, LA-PAM should only be applied when suspended sediment concentrations are greater than 78 mg L⁻¹.

The largest canal to which LA-PAM was applied was the UNC, South Canal (UNC; Table 7-1). Application of LA-PAM began on August 15, 2006, and lasted for 2.3 hours. Samples were collected at a location downstream of the application boundary and at a second location in the Uncompahgre River just downstream of where the South Canal emptied into the river. At the time of the experiment, the South Canal delivered an additional 790 cfs into the Uncompahgre River that was carrying only 350 cfs (measured 5.6 km upstream at USGS Station 09147500). All 50 water quality samples collected were below the detection limit for LA-PAM.

Finally, water samples were collected during the application of emulsified LA-PAM on the Grand Valley Irrigation Company (GVIC; Table 7-1). Application took place between March 27 and 30, 2006, and samples were collected at several locations. For more details as to application and sampling see Section 2.3.2. All samples had LA-PAM concentrations below the detection limit except the furthest downstream site, which had 3.5 mg L⁻¹ of LA-PAM. It is unknown why polymer concentrations were elevated at this site.

![Figure 7-9](Image)

Figure 7-9. LA-PAM concentrations at RFH-1 during application on June 29 to 30, 2006. Samples were collected on the first day at 0.5 and 2.1 km downstream of the application boundary. Operational failure of the boat caused application to last for approximately 9.5 hours. See Section 2.3.7 for more detail.
Figure 7-10. LA-PAM concentrations at CAT-1 during LA-PAM application on June 3, 2006. Application began at 3:30 p.m. and lasted for 2.5 hours. Samples were collected at a single location downstream of the application boundary.

Figure 7-11. LA-PAM concentrations at HID-2 during LA-PAM application on July 16, 2006. Application began at 9:00 a.m. and lasted for 2 hours. Samples were collected at 1,931 and 3,573 m downstream of the application boundary. Data from HID-1 are not presented, as all samples were below the detection limit.

7.2 Acrylamide Monomer

The risk of human health impacts stems from the knowledge that the acrylamide (AMD) monomer is a cumulative neurotoxin and a suspected human carcinogen. Based on an AMD concentration of 0.05 percent in LA-PAM, and the EPA drinking water guideline of 1 mg L\(^{-1}\) for LA-PAM, the AMD drinking water standard has been set by EPA at 0.5 μg L\(^{-1}\).
Even though other forms of PAM have been used extensively in drinking water systems for removal of particulates, in cosmetics as thickeners, in waste water treatment for enhanced settling of precipitated metals, and even as an additive for livestock feed (USFDA, 2006), the risks associated with use of LA-PAM in canal sealing applications have not been completely assessed. The results from field, laboratory, and modeling studies are presented below.

7.2.1 Concentrations of AMD During LA-PAM Application

Acrylamide is only released when the granular polymer is released into water, so a relationship between AMD and LA-PAM concentrations was not unexpected. Laboratory studies indicated an AMD release half-life of about 37 minutes and a total release time of about two hours. During these studies, however, solutions were not as well-mixed, were 10 times more dilute, and were subject to AMD being trapped in partially hydrated LA-PAM as the solution was being filtered. In short, the rate of AMD release is most likely to be dependent on in-canal mixing processes.

Water samples were collected to assess the concentrations of AMD in canal water to which LA-PAM had been added. The number of samples collected was severely limited due to the time-intensive and analytically complex procedure needed to achieve an AMD reporting limit of 0.1 μg L⁻¹. Thirty-six samples analyzed from several LA-PAM application events in 2005 and 2006 indicated that: 1) nearly all samples were less than the U.S. EPA limit of 0.5 μg L⁻¹; 2) elevated AMD concentrations were short-lived and decreased over time; and 3) AMD concentrations appeared to be related to LA-PAM concentrations, which was not unexpected. The majority of samples for AMD analysis were collected during the first four hours of LA-PAM application, when their concentrations were expected to be the highest (Figure 7-12). Of the 36 samples collected, over 50 percent showed AMD concentrations below the detection limit of 0.1 μg L⁻¹. These samples are shown plotting along the lower line in Figure 7-12. As discussed in detail by Young et al (2007a), these concentrations are orders of magnitude below the chronic levels needed to impact human health.

Figure 7-12. AMD concentrations in canal water associated with the application of LA-PAM.
At LAM-1, where the formation of PAM-sediment flocs was not considered to be a dominant process, AMD concentrations remained elevated in conjunction with continually elevated LA-PAM concentrations (Figure 7-13). At LAM-2, AMD concentrations also appeared to mimic those of LA-PAM (Figure 7-14); however, not enough samples were analyzed to verify this trend. During applications at other sites such as SD (Figure 7-15), AMD concentrations were elevated only during the start of LA-PAM application. Higher concentrations were expected near the start of polymer application, as there was little time for AMD or LA-PAM to interact with the environment due to the close proximity of the active polymer application point to the water sampling point. As the time of exposure to canal water increased with the upstream movement of the polymer application point, AMD concentrations decreased. Possible mechanisms for this include AMD dilution with upstream canal water, AMD becoming trapped in PAM-sediment flocs that settle to the bottom of the canal, seepage loss of AMD into groundwater, or utilization by microbes in the canal water. Regardless of the mechanism, it appears that one of the best ways to minimize downstream transport of AMD is to maintain a low or undetectable concentration of LA-PAM in the water column through the application process. This may be accomplished by applying the polymer only under conditions that assure complete use of added LA-PAM through the formation of PAM-sediment flocs.

Figure 7-13. Plot of AMD and LA-PAM concentrations and elapsed time at LAM-1 during LA-PAM application on May 18, 2006. Application lasted for 8.3 hours. Samples were collected at a single location downstream of the application boundary.
7.2.2 Potential Transport of AMD in Groundwater: Field Dye-tracer Study

Field tests were conducted to investigate the potential for AMD transport into groundwater during an LA-PAM application. The experiment was conducted on August 7, 2007, in the Catlin Canal, near Rocky Ford, CO (Figure 7-16), using a dye that was injected into the canal sediments simultaneously with the application of LA-PAM to the canal water. The easily-observable dye was used as a surrogate for AMD, because the number of AMD samples analyzed was limited due to complex sample collection and analytical procedures. Details of field methodology are presented in Section 3.2.
Figure 7-16. Aerial photograph of a portion of CAT showing the location of the tracer test sites, a portion of the canal that was treated with PAM, and the location where the LA-PAM application boat broke down.
Canal water samples taken near the injection well were tested for AMD at both tracer sites (Figure 2; Table 7-2). All six samples remained below the EPA drinking water standard of 0.5 μg L⁻¹. At Site 2, AMD concentrations were above detection limits at both 30 and 60 minutes. In contrast, AMD concentrations were above the detection limits only at 0 minutes at Site 5. Inconsistencies between the two sites are likely due to the breakdown of the LA-PAM application boat, which occurred upstream of Site 5, reducing the amount of LA-PAM and AMD in the water during this time period. Well samples for AMD analyses were chosen based on tracer concentration thought to have the best potential for containing AMD. Unfortunately, the presence of rhodamine interfered with AMD analyses, so these samples could not be analyzed.

Table 7-2. Results of AMD analyses from canal samples collected during the tracer test. Samples containing AMD concentrations below the detection limit are marked as ‘<0.1’, whereas samples with no discernible AMD concentration are marked ‘ND’.

<table>
<thead>
<tr>
<th>Sample ID</th>
<th>Location</th>
<th>Elapsed Time</th>
<th>Concentration μg L⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>4943</td>
<td>Canal Site 2</td>
<td>0 Min</td>
<td>&lt;0.1</td>
</tr>
<tr>
<td>4945</td>
<td>Canal Site 2</td>
<td>30 Min</td>
<td>0.204</td>
</tr>
<tr>
<td>4965</td>
<td>Canal Site 2</td>
<td>60 Min</td>
<td>0.158</td>
</tr>
<tr>
<td>5262</td>
<td>Canal Site 5</td>
<td>0 Min</td>
<td>0.189</td>
</tr>
<tr>
<td>5269</td>
<td>Canal Site 5</td>
<td>30 Min</td>
<td>&lt;0.1</td>
</tr>
<tr>
<td>5276</td>
<td>Canal Site 5</td>
<td>60 Min</td>
<td>ND</td>
</tr>
</tbody>
</table>

Elapsed time is based on when the dye was added to the injection well.

The results of the tracer test were used to investigate the potential transport of AMD from the injection to the observation well. The tests were initiated when two liters of 217,000 mg L⁻¹ rhodamine were poured into each of the injection wells at the beginning of each tracer test. The concentration of rhodamine remained fairly high in the injection well for the first two or three hours of the test then began to decline more rapidly (Table 7-3).

Table 7-3. Results of rhodamine analyses from the injection wells at Sites 2 and 5 on the Catlin Canal.

<table>
<thead>
<tr>
<th>Elapsed Time (Minutes)</th>
<th>Site 2</th>
<th>Site 5</th>
</tr>
</thead>
<tbody>
<tr>
<td>30</td>
<td>&gt;200,000</td>
<td>188,200</td>
</tr>
<tr>
<td>60</td>
<td>&gt;200,000</td>
<td>174,800</td>
</tr>
<tr>
<td>90</td>
<td>&gt;200,000</td>
<td>140,600</td>
</tr>
<tr>
<td>120</td>
<td>&gt;200,000</td>
<td>116,100</td>
</tr>
<tr>
<td>150</td>
<td>198,213</td>
<td>30,000</td>
</tr>
<tr>
<td>180</td>
<td>121,160</td>
<td>27,400</td>
</tr>
<tr>
<td>210</td>
<td>62,033</td>
<td>24,800</td>
</tr>
<tr>
<td>240</td>
<td>29,553</td>
<td>22,100</td>
</tr>
<tr>
<td>270</td>
<td>13,333</td>
<td>8,700</td>
</tr>
<tr>
<td>1380</td>
<td>11,340</td>
<td>3,200</td>
</tr>
<tr>
<td>2760</td>
<td>3,600</td>
<td>1,900</td>
</tr>
</tbody>
</table>
Rhodamine was first observed in the observation well at Site 2 after 90 minutes and not until 1,380 minutes at Site 5 (Table 7-4). The more rapid response at Site 2 was presumably due to preferential flow through sand layers that were noted during the drilling of the wells. Two additional secondary observation wells located 0.67 m (2B) and 1.28 m (2C) away from the injection well showed higher concentrations of rhodamine immediately after injection. Data from secondary observation wells were not used in the analysis due to their high conductivity and close proximity to the injection well.

A thunderstorm and hail stopped routine sample collection and active pumping of the observation wells at both sites during the evening of August 7. The submersible pumps were subsequently activated only during sample collection the following two days. Samples collected both days contained low concentrations of rhodamine (Table 7-4). Although there were insufficient data to determine the maximum rhodamine concentrations in the observation wells at both sites, rhodamine concentrations of less than 6 mg L^{-1} in the observation wells were at least 30,000 times lower than those initially measured in the injection wells. Assuming that AMD was as conservative as the rhodamine tracer, then the 0.20 μg L^{-1} of AMD measured in the canal water would translate to 7 x 10^6 μg L^{-1} of AMD in the groundwater well, approximately 3 m away from the canal. This is over 1400 times lower than the most sensitive laboratory method used to analyze for AMD. If a maximum rhodamine concentration of 1,000 times greater than that measured in the observation wells (e.g., 6,000 mg L^{-1} instead of 6 mg L^{-1}) is assumed, then the resulting estimated AMD concentration is 0.04 μg L^{-1}, still below the 0.1 μg L^{-1} detection limit for AMD.

The actual concentration of AMD in groundwater is likely to be lower than is estimated above. First, it was assumed that AMD in the groundwater immediately under the canal was the same as in the bulk canal water. This is unlikely, as the addition of LA-PAM will also seal the canal, thereby reducing seepage and AMD transfer into the groundwater. Secondly, AMD that does migrate into the soil toward groundwater may be quickly used and degraded by microbial populations (Section 7.2.3). Therefore, based on the results of this and previous sections, it was concluded that the likelihood of measuring AMD concentrations in groundwater wells adjacent to canals treated with LA-PAM is low. However, the connectivity of wells observed at CAT-2 suggests that for safety, LA-PAM should not be applied when groundwater wells are immediately adjacent to the canal

<table>
<thead>
<tr>
<th>Elapsed Time (Minutes)</th>
<th>Concentration (mg L^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Site 2</td>
</tr>
<tr>
<td>90</td>
<td>0.04</td>
</tr>
<tr>
<td>120</td>
<td>0.04</td>
</tr>
<tr>
<td>150</td>
<td>0.11</td>
</tr>
<tr>
<td>180</td>
<td>0.19</td>
</tr>
<tr>
<td>210</td>
<td>0.3</td>
</tr>
<tr>
<td>240</td>
<td>0.37</td>
</tr>
<tr>
<td>270</td>
<td>0.52</td>
</tr>
<tr>
<td>1,380</td>
<td>0.93</td>
</tr>
<tr>
<td>2,760</td>
<td>0.17</td>
</tr>
</tbody>
</table>

Table 7-4. Results of rhodamine analyses from observation wells at Site 2 (2D) and Site 5 (5D). The observation wells were 3.09 m (2D) and 1.98 m (5D) from the injection well. Rhodamine concentrations from samples taken prior to 90 minutes were not detectable (ND) and not reported.
7.2.3 Potential Transport of AMD in Groundwater: Laboratory Studies

The fate and transport of AMD in soil/water systems was studied in the laboratory (Arrowood, 2007). The results of these studies, which are discussed below, indicate: 1) AMD sorption is low and instantaneous, and would not be expected to slow the migration of AMD in soil; 2) AMD concentrations were degraded to below the detection limit of 3 μg L⁻¹ in artificial soil columns within 12 hours of contact time; and 3) initial AMD concentrations in undisturbed canal soils decreased 48 to 59 percent within a 36-hour test period. Based on these results, the AMD half-life in natural systems was estimated to be 30 to 42 hours. Transport models indicated that bacterial degradation of AMD was the primary process and that the dilution of AMD from mixing with uncontaminated canal and groundwater was a secondary process in reducing AMD concentration in groundwater. The general methods used in the following studies are detailed in Section 3.5.3. The studies presented in this section are done so as an overview, specific information regarding the methodology and results can be found in the referenced articles and reports.

The results of the batch experiments showed that AMD sorption was instantaneous and was possibly affected by the fines (clay + silt) content in the soil samples. The mass of AMD that sorbed onto the soil material varied from 1 to 11 percent, 1.6 to 22 percent, and 2.4 to 48 percent for the unsieved #70 mesh sand, unautoclaved loamy sand soil, and the autoclaved loamy sand soil, respectively. Sorption onto sieved #70 mesh sand and C33 sand was not significant. When sorption was expressed as a percentage of the initial concentrations (ranging from 50 to 5,000 μg L⁻¹ of AMD), the results showed a reduced affinity for sorption with increasing concentrations.

Results from abiotic soil column breakthrough experiments were expressed as retardation factors (R) and dispersion coefficients for AMD. The retardation factor is a measure of the sorptivity of a compound to the solid matrix, where R = 1.00 occurs when the solute is nonsorbing and conservative. Retardation values larger than 1.00 indicate that sorption of the compound will occur, hence delaying the arrival of the compound at a particular point in space. The column results for these experiments showed R values very close to, but larger than, 1.00 (mean = 1.02, maximum = 1.07, minimum = 0.98, and coefficient of variation = 2.12 %), indicating that very little sorption should be expected to slow the migration of AMD in soil.

The column experiments were re-executed to examine how microbial breakdown of AMD affected the breakthrough curves. After the C33 sand and loamy sand soil were inoculated with bacteria, significantly higher loss of AMD was observed as the microbial community was able to use the AMD as a food source. Use of AMD occurred even in the presence of competing nitrogen and ammonia sources. Acrylamide concentrations in most tests were, in fact, completely degraded (i.e., at least to below method detection limits) within four pore volumes, or 12 hours of contact time in the soil. Concentrations in the effluent never exceeded 40 percent of the initial concentration. Figure 7-17 shows an example breakthrough curve using C33 sand with naïve bacteria (i.e., not previously exposed to AMD) and with a competing nitrogen source (Labahn et al., 2007). Acrylamide concentrations were reduced after only several hours of exposure (filled circles) compared to the expected breakthrough curve for AMD without microbial degradation (solid line).
Laboratory breakthrough tests were also run on four soil columns filled with undisturbed soil from the RFH. The results showed a 48- to 59-percent reduction of the initial concentration of AMD within the 36-hour test period (Labahn et al., 2007). Comparing AMD transport behavior in both sterile and inoculated columns, the differences in AMD degradation rates and half-lives are substantial. Acrylamide degradation rates in the bacteria columns using C33 sand were about an order of magnitude higher than in the canal column, and the half-lives were 10 times shorter. These differences are primarily due to the higher cell count of bacteria in the bacteria columns versus the undisturbed canal sediments. A four order of magnitude difference existed between these two sources of material. The final estimated half-life of AMD in natural systems was determined to be 30 to 42 hours.

Results from the laboratory experiments were subsequently used as input to HYDRUS-2D (Šimůnek et al., 2006) to simulate the fate and transport of AMD that migrates from a canal prism and into groundwater. Modeling results showed that AMD concentrations depended on the quality of the seal from the PAM treatment. For cases of a full canal seal, or where the canal is underlain by loamy sand soil, AMD entering into the groundwater environment was in very low concentrations and did not migrate to any appreciable distances (i.e., 10 m from the canal). When the canal was only partially sealed by PAM and in soil of #70 mesh sand or C33 sand, conditions more realistic to be found in the field, then bacterial degradation of AMD was the primary factor determining the extent of AMD contamination in the groundwater. Dilution of AMD from mixing with groundwater and with uncontaminated water seeping from the canal played a role in reducing the AMD concentration. Sorption caused only a small delay in the arrival of AMD. If degradation rates
are the same as those measured from soil cores collected from operational canals, then AMD concentrations would be below 7.5 percent of the concentration in the canal water within about 25 m of the canal in all cases tested; i.e., an initial concentration of up to 6.65 µg L\(^{-1}\) would be undetectable 25 m away from the canal. Given that the highest concentration of AMD detected in the field was about 10 times lower (0.65 µg L\(^{-1}\)), the modeling and laboratory results indicate that the likelihood of AMD contamination from LA-PAM treatment of canals –using the methods suggested by Susfalk \(\text{et al.}\) (2007) – would be very small.

### 7.2.4 Description and Brief Summary of Risk Characterization Report Findings

No risk assessment of LA-PAM usage in canals has previously been conducted. To better understand the potential health and environmental effects of LA-PAM usage, a risk characterization based on a literature review of previously published toxicological studies and an evaluation of preliminary data obtained from application in test canals was completed (Young \(\text{et al.}\), 2007a). The following potential risks were identified: (1) ecological and human health impacts associated with the environmental release of residual AMD and (2) the physiological impacts on benthic organisms of LA-PAM release into receiving streams. Two potential pathways – inhalation and ingestion – for AMD exposure as a result of using LA-PAM were identified (Young \(\text{et al.}\), 2007a). Inhalation of AMD was not considered to be likely, due to a very low volatilization potential when it is dissolved in water and the use of dust masks to reduce exposure to airborne AMD particulates during the application process. Ingestion was considered to be more probable, most likely occurring immediately after LA-PAM treatment, through the consumption of canal water and/or groundwater containing AMD. Dermal exposure to LA-PAM was not considered by Young \(\text{et al.}\) (2007a), due to the availability and the recommended use of personal protective equipment by workers applying LA-PAM (Susfalk \(\text{et al.}\), 2007), especially dust masks to reduce exposure to airborne particulate LA-PAM containing AMD during the application process.

Young \(\text{et al.}\) (2007a) examined risk from measured AMD concentrations in canal water samples, and AMD concentrations predicted in treated water based on realistic canal geometries, flow characteristics, and conservative assumptions about LA-PAM hydration and AMD release rates. The concentrations of AMD in water sampled from treated canals were compared to current U.S. EPA drinking water standards, concentrations derived from the lowest doses that caused adverse effects in animal studies, and concentrations equivalent to the highest doses that caused no adverse effects in these studies. Results of laboratory tests conducted on animals were extrapolated to humans using an uncertainty factor of 1,000, as suggested by the U.S. EPA (1988) for the chronic acrylamide oral reference dose (RfD).

Analytical results of samples collected during treatment experiments indicate that observed concentrations were 50 percent below drinking water standards, approximately 1,000 times below the uncertainty factor-adjusted lowest daily doses that caused reproductive impacts in laboratory animals, and about 25 percent of the drinking water concentration associated with a one-in-ten thousand lifetime risk for cancer if consumed daily during a lifetime.

Elevated concentrations of LA-PAM and AMD are expected in surface water samples only during and shortly after treatment (<1 to 12 hours), and in close proximity to the treatment locations. Though residual AMD is released into the canal water when LA-PAM hydrates, the highest concentrations of AMD expected (based on limited field data) are nearly two orders of magnitude below the No-observed-adverse-effect-level (NOAEL) for
human receptor surrogates. Though the limited studies from which data were taken and used in this risk characterization do not span the breadth of potential field conditions that could be expected, the exposure analysis conducted by Young et al. (2007a) indicates that acute (short-term) AMD concentrations in canal waters will be between one and four orders of magnitude below the chronic (long-term) levels needed to impact human health.

During treatment of canals, concentrations of LA-PAM is expected to be similar to, or slightly higher than, those specified by the drinking water standards; however, as spelled out by Susfalk et al. (2007), LA-PAM applications will occur only one to two times per year at a rate of no more than 11.2 kg ha⁻¹ per application. Thus, AMD would be released into the canal water only during these application periods. Additional information on the environmental fate of AMD and LA-PAM is currently being obtained through laboratory and field-based experiments (i.e., Arrowood, 2007; Labahn et al., 2007). However, data collected to date indicate that AMD is readily degraded from microbial activities in both water and soil environments.
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8.0 POTENTIAL RISKS OF LA-PAM APPLICATION II: AQUATIC ORGANISMS

Studies have indicated that protococcal algae, some invertebrates (planaria, scuds [Gammaridae], and *Daphnia* sp.), and fish are affected by the presence of polyacrylamide (Petersen *et al.*, 1987; Beim and Beim, 1994; Muir *et al.*, 1997). However, there is a paucity of information to assess the environmental consequences of its application to aquatic ecosystems and no studies have examined its effect on aquatic macroinvertebrates. To address this deficiency, studies were conducted to examine the response of aquatic macroinvertebrate communities to LA-PAM in an irrigation canal treatment and in a controlled dose-response field experiment using an array of artificial streams supporting a riffle benthic community. The structure of pre- and post-treatment benthic macroinvertebrate (BMI) and drift communities in treated (at a rate of 10 lbs ac\(^{-1}\); 11.2 kg ha\(^{-1}\)) and untreated reaches of the canal were studied. The effect of LA-PAM, applied at a rate of 44.8 kg ha\(^{-1}\) (40 lbs ac\(^{-1}\)), was examined on riffle BMI communities and macroinvertebrate drift during a controlled experiment. Specific results from these studies are presented in Sections 8.1 and 8.2, with a general discussion of the results presented in Section 8.3.

8.1 Smith Ditch BMI Study Results

A total of 94 distinct taxa were identified in substrate samples (mean = 18.6 taxa, range = 9 to 37 per sample) from SD. The community was dominated by midges and other tolerant organisms (mean Hilsenhoff Index = 7.2, range = 5.5 to 8.4). Species richness was highest at Sites 1 and 2 (where it was similar to Kannah Creek), lowest in middle reaches, and moderate in the lowest reaches sampled (Sites 8 and 9) (Figure 8-1). This pattern suggests an active BMI drift from Kannah Creek into upper reaches of the canal, and that the contribution of Kannah Creek drift to lower reaches of the ditch is relatively minor. Density was highest at Site 9 and lowest in the middle reaches, and moderate at Sites 1 and 2 and Kannah Creek (Figure 8-2). Within-site differences between pre- and post-PAM application BMI species richness and density were not statistically significant (Kruskal-Wallis test, \(p > 0.05\)). Within-site Kendall Tau-B coefficients assessing pre- and post-treatment communities ranged from 0.435 to 0.613, which suggests that communities were more dissimilar than indicated by statistical analysis. Although community change may have been relatively small, there was a significant (\(p < 0.05\), two-sample T-test, unequal variances) reduction in the proportion of molluscs (*Physa* sp.) in the Site 9 BMI community following treatment. This suggests that the affect of LA-PAM on community structure may be relatively small but some species may be affected by exposure to LA-PAM.

Canonical correspondence analysis including 17 species examined the influence of the distance of a sample site from Kannah Creek, aquatic vegetation depth, mean water depth, channel width, and exposure to LA-PAM on BMI community structure. Exposure to LA-PAM was the only nonsignificant (\(p > 0.05\)) environmental variable (Figure 8-3), which also suggests that the benthic community is minimally affected by LA-PAM exposure. All canonical axes were highly significant (\(p = 0.001\)) and the first two axes explained 89.1 percent of the species-environment variance (Table 2).
Figure 8-1. Mean (± 2 se) of pre- and post-PAM benthic community species richness in Kannah Creek (KC), and at two control sites (1 and 2) and four sites (3, 4, 8, and 9) receiving LA-PAM treatment in SD for three days before and three days following LA-PAM application during August 2006.

Figure 8-2. Mean (± 2 se) density of pre- and post-PAM BMIs in Kannah Creek (KC), and at two control sites (1 and 2) and four sites (3, 4, 8, and 9) receiving LA-PAM treatment in SD for three days before and three days following LA-PAM application during August 2006.
Figure 8-3. Canonical correspondence analysis biplot of statistically significant environmental factors (p < 0.05) structuring BMI communities in six reaches of SD sampled before and after LA-PAM treatment. DDKC = distance downstream from Kannah Creek, Mnwd = mean water depth, VD = aquatic vegetation depth, CW = channel width.
Table 8-1. Eigenvalues, species, and environmental correlations for CCA analysis of macroinvertebrate communities before and after LA-PAM application of SD. A. BMI communities in control, treatment, and receiving reaches. Total inertia eigenvalues for the first four axes = 0.69. B. Drift communities captured in control and receiving reaches before, during, and following LA-PAM application on August 25, 2005. The total inertia of eigenvalues for the first four axes = 2.080.

<table>
<thead>
<tr>
<th></th>
<th>Axes</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1</td>
</tr>
<tr>
<td>A</td>
<td></td>
</tr>
<tr>
<td>Eigenvalues</td>
<td>0.204</td>
</tr>
<tr>
<td>Spp.-Environment Correlations</td>
<td>0.920</td>
</tr>
<tr>
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<td>29.4</td>
</tr>
<tr>
<td>Cumulative Percent of Variance: Species-Environment Correlations</td>
<td>73.3</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th></th>
<th>Axes</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1</td>
</tr>
<tr>
<td>B</td>
<td></td>
</tr>
<tr>
<td>Eigenvalues</td>
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</tr>
<tr>
<td>Spp.-Environment Correlations</td>
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<tr>
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<td>11.0</td>
</tr>
<tr>
<td>Cumulative Percent of Variance: Species-Environment Correlations</td>
<td>60.2</td>
</tr>
</tbody>
</table>

8.1.1 Pre- and Post-treatment Smith Ditch Drift

The 36 drift samples collected in association with substrate samples captured 2,587 animals including 100 taxa (mean = 12.9, range = 3-28). The proportion of dominant taxa in these samples is shown in Table 8-2. Drift rates (Figure 8-4) and species richness (Figure 8-5) were highest at sites 1 and 2, and both parameters decreased downstream. Differences between sites were statistically significant (p < 0.03, df = 5, Kruskal-Wallis test) for both parameters before and after LA-PAM application. Differences within-site drift rates and richness before and after LA-PAM application were not significant (p > 0.07, two-sample T-test, unequal variances) at any site. Inspection of Figures 8-4 and 8-5 shows that rates and richness were generally lower at treatment sites following LA-PAM application, which contrasts with control sites (1 and 2) where post-treatment drift was greater than before LA-PAM. This may indicate that the drift community was affected by PAM application but the differences were not statistically significant.

Table 8-2. The proportion of dominant taxa in SD drift samples taken in association with BMI substrate samples (A) and samples collected during LA-PAM application (B). Sample sizes were 2,587 for A and 5,236 for B, and dominant species comprised 63 percent and 68 percent of the drift community for A and B, respectively.

<table>
<thead>
<tr>
<th>Taxon</th>
<th>A</th>
<th>B</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baetis notos</td>
<td>0.01</td>
<td>0.06</td>
</tr>
<tr>
<td>Theinemannella</td>
<td>0.09</td>
<td>0.05</td>
</tr>
<tr>
<td>Simulium</td>
<td>0.12</td>
<td>0.05</td>
</tr>
<tr>
<td>Coenagrionidae</td>
<td>0.05</td>
<td>0.04</td>
</tr>
<tr>
<td>Cyprididae</td>
<td>0.04</td>
<td>0.04</td>
</tr>
<tr>
<td>Physa</td>
<td>0.08</td>
<td>0.05</td>
</tr>
<tr>
<td>Pristinella</td>
<td>0.09</td>
<td>0.04</td>
</tr>
<tr>
<td>Copepoda</td>
<td>0.15</td>
<td>0.33</td>
</tr>
</tbody>
</table>
Figure 8-4. Mean (± 2 se) macroinvertebrate drift rates at six sample sites collected three days before and three days following PAM application to Smith Ditch.

Figure 8-5. Mean (± 2 se) species richness of macroinvertebrate communities in the drift at six sample sites collected three days before and three days following PAM application to Smith Ditch.
8.1.2 Smith Ditch During-treatment Drift

Drift samples were collected over a range of LA-PAM concentrations recorded during the August 25, 2005, application to SD (Table 8-3). Chemical analysis showed LA-PAM was present at each site for approximately eight hours (until 00:00 hrs. at site 10), but that maximum concentrations occurred for approximately 1.5 hours near the middle of the exposure period. For the drift analysis, samples were separated into collections made before and after LA-PAM application. The concentration of LA-PAM before application was always zero, but following application it ranged from zero to almost 3 mg L\(^{-1}\). Concentrations were highest at upstream sites (6, 7, and 8) and lower at downstream sites (9 and 10).

Table 8-3. The range in LA-PAM (mg L\(^{-1}\)) and suspended sediment (SSC) (mg L\(^{-1}\)) recorded at each Smith Ditch No. 7 drift sample site during August 25, 2005. Sites 1 and 2 were control reaches and upstream from PAM application.

<table>
<thead>
<tr>
<th>Site No.</th>
<th>PAM</th>
<th>SSC</th>
</tr>
</thead>
<tbody>
<tr>
<td>2</td>
<td>0</td>
<td>ND</td>
</tr>
<tr>
<td>6</td>
<td>0 to 2.33</td>
<td>0 to 48</td>
</tr>
<tr>
<td>7</td>
<td>0 to 2.1</td>
<td>21 to 76</td>
</tr>
<tr>
<td>8</td>
<td>0 to 2.98</td>
<td>9 to 53</td>
</tr>
<tr>
<td>9</td>
<td>0 to 0.99</td>
<td>0 to 34</td>
</tr>
<tr>
<td>10</td>
<td>0 to 1.88</td>
<td>3 to 14</td>
</tr>
</tbody>
</table>

The 113 drift samples collected before, during, and after LA-PAM application captured 5,236 animals including 125 taxa (mean = 9.7, range = 0 – 16). The proportion of dominant taxa in these samples is shown in Table 8-2, and closely resembles the list recorded three days before and following LA-PAM application. Differences between these communities were not significant (\(p > 0.14\), Kruskal-Wallis test, df = 1), but copepods comprised a much greater portion of the community during LA-PAM application than in later and earlier samples. Drift rates (Figure 8-6) and species richness (Figure 8-7) were highest at sites 1 and 2, both parameters were lowest at sites nearest LA-PAM application (sites 6 and 7), and they generally increased downstream through site 10. Drift rates and richness were generally higher during and after LA-PAM application to treatment sites, which is in contrast with the control reaches where both parameters were higher during the period before LA-PAM application. Both of these patterns suggest that macroinvertebrate communities were influenced by LA-PAM exposure. Differences among sites were statistically significant (\(p < 0.05\), df = 5, Kruskal-Wallis test) for drift rates and richness before and after LA-PAM application. Within-site differences in these metrics before and after LA-PAM application were significant only at site 10 (\(p > 0.005\), two-sample T-test, unequal variances).
Figure 8-6. Mean ($\pm$ 2 se) macroinvertebrate drift rates recorded before and after PAM was applied to Smith Ditch No. 7. Sites 1 and 2 (control) were upstream from PAM treatment. The range in PAM concentrations recorded during drift studies at each site is shown in Table 8-3.

Figure 8-7. Mean ($\pm$ 2 se) species richness of drift macroinvertebrate communities before and after PAM was applied to Smith Ditch No. 7. Sites 1 and 2 (control) were upstream from PAM treatment. The range in PAM concentrations recorded during drift studies at each site is shown in Table 8-3.

Canonical Correspondence Analysis (CCA) of drift before and after LA-PAM application to SD showed that (in order of relative importance) suspended sediment concentration (SSC), maximum LA-PAM concentration recorded at a site, period of exposure to LA-PAM, LA-PAM concentration during the sample, and time of day were statistically significant ($p < 0.01$) environmental variables structuring the community (Figure 8-8). First and second canonical axes were also significant ($p = 0.002$), and almost 90 percent of species-environment relationship was explained on the first three axes (Table 8-1). Similarities between the alignment of SSC and maximum LA-PAM concentration along Axis 1 suggest that drift may be influenced by either factor or their potential interacting
influence. This may be attributed to observations during other studies showing that macroinvertebrate drift typically increases during periods of environmental stress, such as exposure to elevated SSC (e.g., Doeg and Milledge, 1991; Bond and Downes, 2003; Wong et al., 2004). Nocturnal and afternoon macroinvertebrate drift rates are generally higher than daytime rates (Bass, 2004), which suggest that drift rates observed during SD studies would naturally increase through the late afternoon and evening. Time of day was a significant environmental variable but its relatively low contribution relative to SSC and LA-PAM suggest that its influence was minor. Close association of time of day and maximum LA-PAM concentration can be attributed to the occurrence of highest LA-PAM concentrations late in the day.

Figure 8-8. Canonical correspondence analysis biplot showing statistically significant environmental variables structuring the macroinvertebrate drift community at sites exposed to PAM treatment in Smith Ditch No. 7 during August 25, 2005. Site numbers as shown in all other figures. B = samples taken before PAM application, A = samples taken after PAM application, PAMC = PAM concentration recorded during each drift sample, SSC = suspended sediment concentration, TMDY = time of day, MPAM = maximum PAM concentration recorded at sample site.

The CCA biplot also shows clustering of most before-treatment communities in the upper left portion of the graph, which indicates that the pre-treatment community at each site was distinctive and that there was little temporal variation in their structure (Figure 8-8). The broad scattering of treatment samples (those exposed to low and high LA-PAM and SSC) indicates that drift communities were influenced by SSC concentration and LA-PAM exposure. The effects were greatest in reaches experiencing highest SSC and LA-PAM concentrations (see Table 4), but they were also evident at sites 9 and 10 where concentrations were low.
8.2 Experimental Troughs

A total of 76 BMI taxa (range 26 to 38) in densities from 5,566 m\(^2\) to 37,200 m\(^2\) were counted in control and treatment troughs. Mean HBI = 6.0 (range = 5.5 to 6.5), which indicates this BMI community is comprised of species less tolerant than the SD community. Suggesting this community may be more susceptible to influences of LA-PAM. Possible effects of LA-PAM treatment were exhibited by significant differences (p < 0.03, paired T-test, unequal variances) between control and treatment densities (Figure 8-9). This differed from studies in SD where BMI density was unaffected by LA-PAM treatment. Treatment effect on species richness was not significant (p > 0.05, paired T-test, unequal variances), but richness in treated troughs was lower than richness in either post-treatment control or pre-treatment troughs (Figure 8-10). This pattern suggests that exposure to LA-PAM influenced this BMI community, but the effect was not statistically significant.

Figure 8-9. Mean (± 2 se) density of substrate BMI communities occurring in experimental control and treatment troughs before and during PAM application at 40 lbs/acre.

Figure 8-10. Mean (± 2 se) species richness in substrate BMI communities occurring in experimental control and treatment troughs before and during PAM application at 40 lbs/acre.
A detrended correspondence analysis (DCA) also indicated that the substrate community was affected by LA-PAM treatment (Figure 8-11). With the exception of the before-treatment community in trough number 8 (PB8), all untreated communities were broadly clustered in the upper portion of the DCA plot and treated communities were clustered along a line that was below this grouping. The distant placement of PB8 from all other treated and nontreated communities suggests this community was unusual and an outlier, which may have little relevance to the analysis.

![Figure 8-11. Deterend correspondence analysis plot of BMI substrate communities in control (C) and treatment (P) troughs before (B) and four days following (A) PAM application at 40 lbs/acre. Treated trough communities are identified by triangle. Diagonal line indicated division between untreated (above) and treated communities (below) (with exception of trough PB8, which is interpreted as an outlier with little relevance to this analysis).](image)

Differences in drift rates and species richness before and during treatment were not significant ($p > 0.22$, paired T-test, unequal variances), but drift rates were generally greater during treatment than before (Figure 8-12). A DCA plot shows pre- and during-treatment drift communities were relatively similar, but treatment communities were clustered in the lower center of the plot, and mostly segregated from control and pre-treatment communities (Figure 8-13). Communities in troughs 8 and 2 appeared to be most affected by treatment, and the effect to the trough 11 community was minimal. The control sample from trough 9 during treatment also appears to be an outlier and may be relatively unimportant to this analysis.
Figure 8-12. Mean (± 2 se) number of animals captured in 30 minutes of drift from experimental control and treatment troughs before and during PAM application at 40 lbs/acre.

Figure 8-13. Detrended correspondence analysis plot of drift communities captured from control (C) and treatment (P) troughs before (B) and during (D) PAM application at 40 lbs/acre. Treated troughs are identified by triangle.

Treatment appeared to affect *Baetis tricaudatus* (a mayfly) and *Simulium* sp. (a blackfly) more than other species. Drift rates of both species were higher in treatment than control troughs, but these differences were not significant (p > 0.22, paired T-test, unequal variances) (Figure 8-14). This suggests that these species may be more sensitive to LA-PAM than other species, but their response was insufficient to alter the structure of a community in a treated section of the canal. Similar to observations in SD, it also appeared that drift activity of some species increased during treatment.
8.3 Discussion of the Effects of LA-PAM Application on Aquatic Organisms

Irrigation canals are relatively harsh, low quality-aquatic systems. The SD is an appropriate site to examine the effects of LA-PAM on aquatic organisms in irrigation canals because, like other western U.S. irrigation canals, it is dried annually, and its riparian and instream vegetation are mechanically removed to maintain the ditch. The LA-PAM application at SD was characterized by prolonged exposure (approximately 8 hours) to relatively low LA-PAM concentrations, which is consistent with conditions that may occur during a ‘typical’ canal application of LA-PAM.

Aquatic BMI communities are structured by many natural and human factors. Relationships between the structure of BMI communities and factors influencing stream environment have been examined in a number of studies (e.g., Poff and Ward, 1989; Barreiro and Pratt, 1994; Clements, 1999) and a number of metrics have been developed to assess environmental harshness in context of BMI community structure.

The HBI is one numerical metric used to quantify the gradient of environmental harshness and corresponding tolerance of the BMI community. Communities in high-quality habitats are dominated by BMIs that are intolerant of harsh conditions (e.g., high temperature and turbidity, fine substrate, high nutrient and low dissolved oxygen concentrations). An HBI less than 4 is indicative of these environments. As environmental harshness increases, intolerant species are replaced by taxa that tolerate harsh conditions; conversely, an HBI greater than 8 is indicative of these communities. At SD, the HBI for its BMI community was 7.2, indicating that it was tolerant of very harsh conditions.

When exposed to stress, BMIs release from the substrate and drift downstream, and drift is an indicator of BMI stress (Gibbins et al., 2005). Macroinvertebrates quickly respond to changes in environmental conditions, such as environmental disturbance or pollution events (e.g., Clements, 1994, 2000) and changes in community composition may occur because of mortality from lethal effects or because of exposure to unsuitable conditions (e.g., Doeg and Milledge, 1991; Bond and Downes, 2003). For instance, experimental
application of pyrethroid pesticides caused an immediate and significant increase in drift rates of BMIs in mesocosms (Heckmann and Friberg, 2005), which lead to altered community composition of the exposed community relative to controls.

Neither BMI density nor community structure appeared to be affected by LA-PAM application to SD, although the number of molluscs decreased substantially following treatment at one site. This suggests that LA-PAM may have little effect on tolerant BMI communities that inhabit irrigation canals, though individual species (such as the mollusk) may be affected by exposure to LA-PAM. There was little difference in macroinvertebrate drift during days before and following LA-PAM treatment. However, during the days after treatment, rates and species richness at treated sites were consistently lower, while rates and richness at control sites were higher. These observations may suggest that post-treatment macroinvertebrate drift was depressed for several days following LA-PAM application.

At SD site 10, the effect of LA-PAM on drifting macroinvertebrates during treatment was less equivocal. Increases in drift rate and richness during application were statistically significant at this site and CCA showed that LA-PAM concentration was a significant factor influencing drift community structure. Although not statistically significant, species richness and drift rates at most sites were also higher following LA-PAM application. Increases in these metrics observed at treatment sites contrasted with observations at control sites where drift rates before application were greater than during periods after it was applied. Differences in drift patterns observed during treatment studies and drift studies conducted days before and following treatment may be attributed to the downstream transport of large numbers of organisms during and immediately following LA-PAM treatment. Treatment may have greatly reduced the number of drifting organisms remaining within the treated reach, which decreased drift rates for at least three days following LA-PAM application.

Trough experiments involved a less tolerant BMI community (HBI = 6.0) (which is more indicative of a community occupying a natural system) and exposure to a high LA-PAM concentration for a very short period (less than 1 hour). From these studies, substrate BMI density and species richness decreased following treatment. Density differences were significant, richness differences were not, but richness following treatment was lower than before treatment. A DCA examining the BMI community indicated that structure of treated communities also differed from control communities. Drift rates from control and treatment troughs were not significant, but rates were generally higher in treatment than in control troughs. There was no apparent difference in the richness from control and treatment troughs. This was largely confirmed by DCA of the drift community, but treatment communities were more similar to one another than to communities in control and pre-treatment troughs. Experimental results indicated that some species are more sensitive to LA-PAM than others, which is congruent with SD observations showing that the effect of LA-PAM was greater on molluscs and copepods than other species. Increased drift of *Baetis tricaudatus* and *Simulium* sp. during LA-PAM treatment is consistent with studies finding that *Baetis* spp. and *Simulium* sp. drift is disproportionately high relative to other, more sessile taxa during stressful conditions (Bass, 2004; Wong *et al.*, 2004).

Most differences between pre- and post-LA-PAM treatment were statistically insignificant in these studies, but SD and trough studies both documented response patterns and statistically significant results showing that macroinvertebrates are adversely affected by LA-PAM exposure. Patterns describing relationships between the environment and the
abundance and distribution of species are readily indicated when results are statistically significant, but statistical significance is rarely observed in studies examining BMI communities because of high variability in spatial and temporal variation (Vinson and Hawkins, 1996). These difficulties do not indicate diminished rigor of these studies, because it is important to assess response patterns and statistical significance to adequately assess the relationship of BMI communities and their environment. While statistical tests provide a framework for analysis, it may be difficult to discern the ecological relevance of statistically insignificant comparisons that may be provided by examining response patterns. This work examined only LA-PAM as the proximate factor affecting BMIs. Studies are needed to determine ultimate factors, which may be physiological or ‘mechanical’ (pathways that influence metabolism by physically blocking oxygen transport across gills). This work also provides little insight into how exposure concentrations or duration may affect BMIs. Environmental consequences of using LA-PAM in irrigation systems are relatively minor because these are low-quality aquatic systems that are frequently disturbed by vegetation management and drying. However, effects observed in these studies suggest that the consequences of releasing LA-PAM into natural aquatic systems are likely to be detrimental, which suggest that it should not be applied under circumstances where this may occur.
9.0 FACTORS AFFECTING LA-PAM APPLICATION

Numerous factors and their interrelationships affect the rate at which granular LA-PAM hydrates, its ability to react with suspended solids, the extent to which it remains in the water column, and its availability to be transported downstream. These factors include the SSC, the concentration and composition of dissolved solids in the canal water, water temperature, application method, and the rate and total mass of granular LA-PAM applied to the canal (physicochemical factors affecting hydration are discussed in Section 4.1). Laboratory studies were conducted to gain a better understanding of how these factors affect flocculation, and the subsequent ability of LA-PAM to reduce seepage rates. Although results from laboratory studies may not always be directly transferable to field conditions, they provide the controlled environment necessary to study the various phases that LA-PAM goes through after it is applied to the canal water.

The first part of this chapter (Section 9.1) details three laboratory studies. In summation, results from these studies indicate that the formation of PAM-sediment flocs are: 1) optimized when LA-PAM concentrations are 1 to 2 mg L⁻¹ in the water column and that higher polymer concentrations provide little additional benefit; 2) limited at low SSC; 3) suppressed at low levels of total dissolved solids levels (≤100 mg L⁻¹); and that 4) LA-PAM typically hydrates under field conditions ranging between 24 and 34 min at 5.1 and 24.4°C, respectively.

The second part of this chapter (Section 9.2) presents a discussion of how the processes observed in the laboratory studies impacted the field studies described in this report. Several important conclusions were drawn. First, the variation in SSC was more critical to efficacy than variations in water chemistry. Second, the augmentation of SSC by dredging or adding sediment should be avoided. Third, the development of the clear zone can be used as a diagnostic tool for evaluating risks and benefits. Fourth, elevated LA-PAM concentrations were primarily found to be associated with the full development of a clear zone as SSC sources become exhausted. Last, the use of application rates based on the wetted perimeter of the canal results in over-application to small-volume canals and possible under-application to larger-volume canals. Instead, application rates should be based on the mass of suspended sediments.

9.1 Laboratory Studies

9.1.1 LA-PAM Concentrations and SSC

The addition of LA-PAM to irrigation canals is not effective without the presence of suspended solids. The polymer reacts with suspended solids in the water column to create the PAM-sediment flocs that ultimately settle to the bottom of the canal and reduce seepage.

A series of jar-test studies were conducted to assess the influence of LA-PAM and SSC on PAM-sediment floc formation. Jar-test methodology is routinely used to assess flocculation and settling characteristics of different polymer systems in the drinking water industry. In these studies, flocculation was assessed using continuous turbidity measurements to determine the degree to which suspended sediment was removed from the water column as detailed in Section 3.5.2. It follows that treatments with a greater percent turbidity reduction were more effective at reducing the SSC by the formation of PAM-sediment flocs. Under the laboratory conditions presented here, the natural drop in turbidity of between 6 and 10 percent increased to 37 to 72 percent with the addition of LA-PAM (Figure 9-1).
Figure 9-1. Impact of LA-PAM addition on SSC. Experiments were run in 0.001 M CaSO₄ with kaolinite used as the suspended solids source. Additional experiments at 600 mg L⁻¹ of suspended solids and all LA-PAM concentration levels were conducted but not included in the figure, as they were similar to the results at 300 mg L⁻¹. Error bars, where present, represent the standard error of at least three replicate runs.

An analysis of variance (ANOVA) indicated that the extent of turbidity reduction was due to both LA-PAM concentration \( (p = 0.0194) \) and the suspended solids concentration (SSC: \( p \leq 0.0001 \)). Post-hoc Bonferroni mean comparisons indicated that the use of LA-PAM significantly increased turbidity reduction but that there was no overall difference between the LA-PAM concentrations used. However, student’s t-test comparisons between individual treatments showed that higher LA-PAM concentrations \( (\geq 8 \text{ mg L}^{-1}) \) reduced PAM-sediment floc formation at lower sediment concentration levels \( (\leq 150 \text{ mg L}^{-1}) \) (Figure 9-1). Higher LA-PAM concentrations result in the polymer chain remaining coiled or in weakly extended conformations, which sterically hinder its reaction with suspended solids (Misra, 1996). Flocs will settle more slowly due to the higher viscosity of the matrix solution. Based on these results, it can be concluded that the optimal concentration of LA-PAM is between 1 and 2 mg L⁻¹ and that higher concentrations do not significantly improve floculation. These findings are consistent with other studies investigating anionic polyacrylamides (McLaughlin and Bartholomew, 2007).

For SSC, the reduction in percent turbidity \( (%\text{TR}) \) increased linearly between 0 and 150 mg kaolinite L⁻¹ \( (%\text{TR} = 0.0027 \times \text{SSC} + 0.0134; R^2 = 0.99) \). However, there was little change in turbidity reduction between 150 and 600 mg kaolinite L⁻¹. Therefore, under these laboratory conditions, an SSC of about 150 mg L⁻¹ is necessary to provide the optimum reduction in turbidity from reaction with LA-PAM. Concentrations less than 150 mg L⁻¹ result in less effective PAM-sediment floc formations. Although higher concentrations of suspended sediment did not improve the relative percent turbidity reduction metric, the total quantity of suspended sediment will be greater at higher sediment concentration levels. For example, given a 60 percent turbidity reduction rate, 180 mg L⁻¹ of suspended sediment will settle with a 300-mg L⁻¹ starting level compared to 360 mg L⁻¹ of sediment will settle with an
initial starting level of 600 mg L\(^{-1}\). This example is simplified, as the relationship between turbidity (an optical measurement) and suspended sediment (a physical measurement) likely changes with local polymer concentrations. This change, however, is not likely of sufficient magnitude to greatly impact this example. It is currently unknown what impact a heavier PAM-sediment floc will have on seepage reduction in canals, but it is likely that a heavier settled floc could result in greater seepage reductions and/or augment the length of time that the seepage remains reduced.

Jar tester experiments indicated that the speed of turbidity reduction was faster in treatments with higher suspended sediment concentrations. The relative rate that turbidity was reduced in a treatment was assessed using an index equal to the slope of the inflection point (SIP) of the curve of turbidity level over time. In Figure 9-2, treatments with lower 10\(^{\text{SIP}}\) values indicated faster reductions in turbidity due to LA-PAM application. A lower SIP index was always correlated with faster elapsed times to the SIP index; however, only elapsed times between individual LA-PAM levels can be compared due to experimental artifacts. For example, at 1 mg L\(^{-1}\) LA-PAM, the time to the SIP decreased from 560 s at 150 mg L\(^{-1}\) to 290 s at 600 mg L\(^{-1}\) of suspended sediment concentration. There is a greater likelihood for interaction between the PAM chain and suspended sediment at higher sediment concentrations resulting in the PAM-sediment floc becoming heavier faster than at low concentrations. In canal applications, a faster settling would result in a shorter distance between where LA-PAM is applied and where it effectively begins to reduce seepage.

![Figure 9-2: Slope of the inflection points (SIP) for data presented in Figure 9-3. The SIP is an index of how quickly flocculation and settling occur after LA-PAM addition. The SIP data were power transformed prior to a one-way qualitative ANOVA analysis. Lower values of 10\(^{\text{SIP}}\) resulted in a faster reduction in turbidity. Treatments with the same letter are not significantly different (a=0.05) as determined by a post-hoc Bonferroni test.](image)

9.1.2 Water Chemistry

The jar tester methodology was used to investigate the impacts of concentration and composition of total dissolved solids (TDS) on PAM-sediment floc formation (Figure 9-3). Results of this laboratory study found that low TDS, in the range of 100 mg L\(^{-1}\), reduces the formation of PAM-sediment flocs, and that low calcium solutions (low SAR) only impacts floc formation at low TDS levels.
Figure 9-3. Role of TDS and SAR on PAM-sediment floc formation. The SAR is an index of the amount of cationic charge contributed to by monovalent sodium relative to that contributed by divalent calcium and magnesium ions. The SAR value will increase with a greater contribution by sodium cations. This experiment was run using the jar tester methodology with 1.5 mg PAM L⁻¹ and 225 mg kaolinite L⁻¹ held constant across all runs. Error bars (when present) represent the standard error of replicate runs.

A qualitative ANOVA revealed that a total TDS concentration of 100 mg L⁻¹ resulted in a significantly lower (50±5 %; mean ± standard error) turbidity reduction compared to all other treatments (83 ± 1 % at 600 mg L⁻¹, 88 ± 1 % at 1,100 mg L⁻¹, and 91 ± 1 % at 1,600 mg L⁻¹). These results are consistent with others that found that cation bridging between anionic polymers and sediment was improved in solutions having a higher concentration of divalent cations (Aly and Letey, 1988; Ben-Hur et al., 1992; Lu et al., 2002). Laboratory results also indicated that a manipulation of monovalent to divalent cations by adjusting the SAR had no significant overall effect on turbidity reduction. However, PAM-sediment floc formation was suppressed at the lowest TDS level (100 mg L⁻¹) with either an SAR of 0 or greater than or equal to 3. Higher SAR values indicate a greater contribution of monovalent to divalent cations, which would be expected to reduce the effectiveness of LA-PAM in soil environments (Lentz and Sojka, 1996). The suppression of flocculation in a dilute calcium sulfate (SAR = 0) at 100 mg L⁻¹ TDS suggests that the presence of some small amounts of sodium (SAR = 0.1 and 1.0) promotes PAM-sediment flocculation in low TDS solution. It should also be noted that the physical processes of random mixing might play a larger role on the ability of LA-PAM and suspended sediment to react with each other under nonoptimum conditions. This was suggested by the greater variability in results for all treatments at the 100 mg L⁻¹ level compared to all other TDS levels.

For canal systems, these results suggest that canal waters containing higher TDS levels are better at promoting the formation of PAM-sediment flocs. Although turbidity reduction of the suspended kaolinite solutions was not very sensitive to changes in SAR,
other reports suggest that soil mineralogy and texture can play an important role in promoting flocculation with polyacrylamides (Peng and Di, 1994; Laird, 1997; Lu et al., 2002; McLaughlin and Bartholomew, 2007).

9.1.3 Influence of Water Temperature on PAM Hydration

Commercial crystalline LA-PAM is capable of absorbing water volumes many times its mass through a series of chemical reactions. Within one hour, LA-PAM is normally dissolved completely. However, variations in water temperature play an important role in the exact amount of time needed to achieve total hydration of PAM. Polyacrylamide hydration time measurements in DI water and in 0.005-M calcium sulfate solutions are reported here at a number of different water temperatures in the approximate range 5 to 30°C. We assumed that hydration time was related to solution viscosity, which would increase and then reach an approximate constant value. The results presented below show that the hydration of LA-PAM in DI water and in 0.005-M calcium sulfate solutions averages about 28 to 29 minutes at 15.09°C.

Hydration time was defined as the time when the viscosity positive slope flattened out and the viscosity no longer changed with time (Figure 9-4). Due to local variations in viscosity during hydration, as well as agitation of the solutions, laboratory experiments resulted in clusters of data points instead of the idealized curve shown in Figure 9-4. Hydration times were estimated from visual examination of the data cluster and through analysis of trend lines in Microsoft Excel®. Where possible, linear regressions were generated separately for the positive slope cluster and the plateau cluster and the two equations were set equal to each other, rearranged, and solved for hydration time (Figure 9-5).

The regression relationships for the DI water and calcium sulfate solutions were essentially the same (Figure 9-5). However, a difference between the DI water and calcium sulfate solutions was observed during the viscosity measurements, when the DI water solutions routinely showed a greater increase in solution viscosity. About 7 to 8 mg L⁻¹ PAM was sufficient to increase viscosity in the DI water experiments, but twice that amount was needed in the salt solutions to achieve a clearly measurable increase in solution viscosity. Cations such as the calcium ions in the salt solution have been shown to inhibit total hydration of some PAM hydrogels (Bowman and Evans, 1991), which may explain the relatively lower viscosity readings for PAM in the calcium sulfate solutions. Also, due to interactions between partially hydrated PAM and the viscometer probe, viscosity values varied over a wider range for the DI water solutions, making it necessary to process numerous samples to obtain usable data sets. This problem was either minimal or absent for the calcium sulfate solutions.
Figure 9-4. Complete hydration is reached where the discontinuity in the viscosity versus time curve occurs.

Figure 9-5. PAM hydration time vs. temperature for DI water (A) and 0.005-M calcium sulfate solutions (B).

Despite the challenges associated with using the vibratory viscometer, strong linear correlations resulted. The equations in Figure 9-5 can be used to calculate PAM hydration time for any water temperature that might typically occur in an irrigation canal, assuming that turbulent agitation in our test vessel was similar to turbulent flow in a canal. For example, using the correlation from Figure 9-5B, typical field temperature data gave hydration times in the range of about 24 to 34 min (5.09 to 24.35°C), with a mean value of about 28 to 29 minutes (15.09°C). Hydration times were essentially the same for the polymer in DI water and in calcium sulfate solutions because hydration was determined by temperature-dependent water diffusion into the polymer. These values agree with previous studies of diffusion of PAM and other polymers similar to PAM (Kishimoto and Kitahara, 1964; Barrie, 1968; Bowman and Evans, 1991).

9.2 Discussion of How Environmental Factors Affected LA-PAM Applications

The success of an LA-PAM application to an unlined water delivery canal must be measured both in terms of seepage reduction and the minimization of environmental impacts. In planning an application, three sequential questions must be addressed. First, is the water quality in the canal conducive to forming PAM-sediment flocs? Second, is there enough
suspended sediment in the canal water to form PAM-sediment flocs? Third, based on environmental conditions, is the application rate and approach sufficient to minimize the downstream transport of hydrated LA-PAM?

9.2.1 Amount and Type of Total Dissolved Solids

Laboratory studies discussed above indicate that the ratio of monovalent to divalent cations in solution becomes important in water having low TDS ($\leq 100$ mg L$^{-1}$). Two observations were made based on the ambient water chemistry of field polymer applications conducted in 2006 and 2007 (Table 9-1). First, the ability to form PAM-sediment flocs was expected to be suboptimal at HID-1, HID-2, and UNC experiments due to TDS concentrations of less than 160 mg L$^{-1}$. Seepage rates at HID-1 were not reduced despite the formation of PAM-sediment flocs indicated by decreasing turbidity levels after polymer application. However, a large concentration of calcium and magnesium may not have been needed to form cation bridges during HID-1 and HID-2 experiments due to low levels of suspended sediment at HID. At UNC, seepage reduction was not estimated due to the low native seepage rates measured prior to application, but in-situ turbidity monitoring did show that turbidity remained consistent, indicating that PAM-sediment flocs did not readily form. The second observation is that flocculation ability did not appear to be affected by SAR, as those sites characterized by low TDS also had low SAR values. As noted by the SAR laboratory study, however, the importance of SAR to flocculation is likely specific to the sites and soils involved, and so may be more important at sites other than the seven studied here.

Table 9-1. Water chemistry from various water delivery canals. See Table 2-2 for site locations. PAM was applied at all locations except at GVIC and KID. TSS is total suspended sediment, DOC is dissolved organic carbon, TDS is total dissolved solids, and SAR is sodium adsorption ratio. LA-PAM was not applied to two other canal reaches along the Grand Valley Irrigation Company (GVIC) in Grand Junction, CO or the Klamath Irrigation District’s (KID) B-canal.

<table>
<thead>
<tr>
<th>#</th>
<th>Location</th>
<th>Exp</th>
<th>Date</th>
<th>EC (μS cm$^{-1}$)</th>
<th>pH</th>
<th>Tu (NTU)</th>
<th>TSS (mg L$^{-1}$)</th>
<th>DOC (mg L$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Catlin</td>
<td>CAT-1</td>
<td>06/03/06</td>
<td>674</td>
<td>7.7</td>
<td>215</td>
<td>471</td>
<td>2.2</td>
</tr>
<tr>
<td>2</td>
<td>Catlin</td>
<td>CAT-2</td>
<td>08/07/07</td>
<td>787</td>
<td>7.9</td>
<td>1,165</td>
<td>817</td>
<td>4.55</td>
</tr>
<tr>
<td>3</td>
<td>GVIC</td>
<td></td>
<td>08/21/06</td>
<td>888</td>
<td>8.2</td>
<td>46</td>
<td>102</td>
<td>2.52</td>
</tr>
<tr>
<td>4</td>
<td>GVIC</td>
<td></td>
<td>09/12/06</td>
<td>896</td>
<td>8.0</td>
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<td>308</td>
<td>2.42</td>
</tr>
<tr>
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<td>GVIC</td>
<td></td>
<td>10/30/06</td>
<td>863</td>
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<td>66</td>
<td>209</td>
<td>2.41</td>
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<tr>
<td>6</td>
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<td>HID-1</td>
<td>07/14/06</td>
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<td>7.7</td>
<td>56</td>
<td>77.5</td>
<td>1.63</td>
</tr>
<tr>
<td>7</td>
<td>Huntley</td>
<td>HID-2</td>
<td>07/16/07</td>
<td>242</td>
<td>7.8</td>
<td>19</td>
<td>31.0</td>
<td>NA</td>
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<tr>
<td>8</td>
<td>KID</td>
<td></td>
<td>08/30/06</td>
<td>142</td>
<td>6.9</td>
<td>5</td>
<td>221</td>
<td>8.28</td>
</tr>
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<td>Lamar</td>
<td>LAM-1</td>
<td>05/19/06</td>
<td>2,570</td>
<td>8.1</td>
<td>8</td>
<td>10.8</td>
<td>3.58</td>
</tr>
<tr>
<td>10</td>
<td>Lamar</td>
<td>LAM-2</td>
<td>06/07/06</td>
<td>2,140</td>
<td>7.9</td>
<td>60</td>
<td>155</td>
<td>3.18</td>
</tr>
<tr>
<td>11</td>
<td>RFHL</td>
<td>RFH-1</td>
<td>06/30/06</td>
<td>512</td>
<td>7.6</td>
<td>68</td>
<td>144</td>
<td>2.19</td>
</tr>
<tr>
<td>12</td>
<td>RFHL</td>
<td>RFH-2</td>
<td>07/20/06</td>
<td>--</td>
<td>--</td>
<td>--</td>
<td>670*</td>
<td>--</td>
</tr>
<tr>
<td>13</td>
<td>RFHL</td>
<td>RFH-4</td>
<td>08/06/07</td>
<td>674</td>
<td>--</td>
<td>--</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
<td>14</td>
<td>UNC</td>
<td>UNC</td>
<td>08/15/06</td>
<td>224</td>
<td>7.6</td>
<td>10</td>
<td>27</td>
<td>2.25</td>
</tr>
</tbody>
</table>

*Suspended sediment estimated from real-time turbidity readings.
Table 9-1. Water chemistry from various water delivery canals (continued).

<table>
<thead>
<tr>
<th>#</th>
<th>Location</th>
<th>TDS</th>
<th>SAR</th>
<th>Ca</th>
<th>Na</th>
<th>Mg</th>
<th>HCO₃</th>
<th>Fe</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Catlin</td>
<td>443</td>
<td>2.3</td>
<td>37.5</td>
<td>71.2</td>
<td>21.7</td>
<td>150</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>2</td>
<td>Catlin</td>
<td>532</td>
<td>1.2</td>
<td>83.3</td>
<td>47.95</td>
<td>25.05</td>
<td>157</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>3</td>
<td>GVIC</td>
<td>519</td>
<td>2.8</td>
<td>64.2</td>
<td>96.5</td>
<td>16.4</td>
<td>163</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>4</td>
<td>GVIC</td>
<td>525</td>
<td>2.8</td>
<td>64.0</td>
<td>97</td>
<td>16.0</td>
<td>164</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>5</td>
<td>GVIC</td>
<td>513</td>
<td>2.6</td>
<td>64.6</td>
<td>91.1</td>
<td>17.6</td>
<td>178</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>6</td>
<td>Huntley</td>
<td>155</td>
<td>0.6</td>
<td>21.9</td>
<td>12.9</td>
<td>6.69</td>
<td>89.5</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>7</td>
<td>Huntley</td>
<td>142</td>
<td>0.6</td>
<td>21.8</td>
<td>13.5</td>
<td>7.04</td>
<td>89.7</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>8</td>
<td>KID</td>
<td>125</td>
<td>0.7</td>
<td>8.99</td>
<td>11.2</td>
<td>4.83</td>
<td>61.6</td>
<td>0.05</td>
</tr>
<tr>
<td>9</td>
<td>Lamar</td>
<td>2,520</td>
<td>3.8</td>
<td>234</td>
<td>294</td>
<td>128</td>
<td>222</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>10</td>
<td>Lamar</td>
<td>1,740</td>
<td>2.9</td>
<td>189</td>
<td>196</td>
<td>89.3</td>
<td>184</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>11</td>
<td>RFHL</td>
<td>297</td>
<td>0.8</td>
<td>50</td>
<td>26.2</td>
<td>14.8</td>
<td>111</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>12</td>
<td>RFHL</td>
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<td>--</td>
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<td>--</td>
<td>--</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
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<td>RFHL</td>
<td>440</td>
<td>1.1</td>
<td>70.2</td>
<td>40.6</td>
<td>22.7</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
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<td>UNC</td>
<td>135</td>
<td>0.2</td>
<td>29.8</td>
<td>5.23</td>
<td>3.04</td>
<td>96.4</td>
<td>&lt;0.02</td>
</tr>
</tbody>
</table>

9.2.2 Suspended Sediment Concentrations

Perhaps the most important factor of all those discussed is the SSC in canal water, as it is the other major component besides the polymer that is needed for the formation of PAM-sediment flocs. Laboratory studies have indicated that between 150 and 300 mg L⁻¹ of suspended sediment is necessary to maximize the formation of LA-PAM sediment flocs based on application rates of 11.2 kg ha⁻¹ (10 lbs ca⁻¹). Field studies that had low levels of suspended sediment (HID-1, HID-2, LAM-1, and UNC) were found to either have no reduction in seepage rate or not readily form PAM-sediment flocs, as discussed above.

Manually increasing suspended sediments concentrations during LA-PAM application does not necessarily increase the effectiveness of an application event and should be avoided for several reasons. First, the location of sediment addition relative to polymer addition must be carefully considered to ensure that sufficient sediment remains in suspension to be available after LA-PAM has hydrated. High LA-PAM concentrations observed during the KC-1 experiment were likely due to the settling of the sediment prior to the effective hydration of LA-PAM. Second, sediment should not be dredged from the bottom of the canal, as disturbance of the canal bed may result in increased seepage rates. Third, sediment sourced from the canal bed (either immediately or during past canal cleanings) will have a larger mean particle size diameter than sediment suspended in the water column. Coarser sediment is less effective at forming PAM-sediment flocs and will not stay suspended in the water column for very long. These latter two factors may explain the lack of PAM-sediment floc formation and the increased seepage rates found after LA-PAM application at the LAM-1. Fourth, manual sediment addition achieves neither the elevated concentration nor the duration of time needed to be effective. In the studies conducted here, manual addition of suspended sediment yielded concentrations that were orders of magnitude lower than those occurring naturally during storm events. Fifth, sediment addition may require a discharge permit and elevated suspended sediment levels may adversely affect downstream users and aquatic habitats. Finally, sediment addition is typically limited to small canals, where water volumes are low and the amount of added sediment is manageable.
The addition of sediment to large canals is prohibitive due to the amount of sediment needed for an entire treatment reach. At some locations, such as in canals downstream of reservoirs that do not experience elevated SSC, there may be not other alternative. In these cases, the guidelines presented by Susfalk et al. (2007) should be used in addition to potentially scaling back the polymer application rate to match the level of suspended sediment in the canal (discussed below).

Unlike other environmental factors that affect LA-PAM efficacy, SSC can vary dramatically during the irrigation season. Therefore, the best option may be to delay application of LA-PAM until a naturally occurring rise in canal SSC occurs, such as following a rain event. Though suspended sediment is elevated during early-season runoff, the colder temperatures and faster velocities typical of this water make it less desirable for LA-PAM application.

9.2.2.1. Implications of clear zone development

The clear zone develops when the concentration of suspended solids remaining in the water column is reduced due to the settling of PAM-sediment flocs to the canal bottom. Based on the results presented in this report, the clear zone can be used as a diagnostic feature to quickly assess the effectiveness of floc formation and the potential downstream transport of hydrated LA-PAM. (Table 9-2). The results presented in Table 9-2 are supported by observations from field and laboratory studies.

Table 9-2. Diagnostic features of the clear zone development.

<table>
<thead>
<tr>
<th>Extent of Clear Zone Development</th>
<th>Potential Seepage Reduction</th>
<th>Potential Risk of Availability and Downstream Transport</th>
<th>Result</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>No Development</strong> (No change in turbidity)</td>
<td>Low</td>
<td>High</td>
<td>PAM-sediment flocs did not form.* Suspended sediment and dissolved solids (cations) may not be sufficient to promote flocculation.</td>
</tr>
<tr>
<td><strong>Some Development</strong> (Small drop in turbidity)</td>
<td>Low to Moderate</td>
<td>Low</td>
<td>PAM-sediment flocculation occurred; however, LA-PAM application rate may be too low.</td>
</tr>
<tr>
<td><strong>Partial Development</strong> (Significant drop in turbidity)</td>
<td>High</td>
<td>Low</td>
<td>Significant PAM-sediment flocculation occurred. Excess suspended sediment remaining in the water column reduces the likelihood of downstream LA-PAM transport.</td>
</tr>
<tr>
<td><strong>Full Development</strong> (Water becomes clear or nearly clear)</td>
<td>High</td>
<td>Moderate to High</td>
<td>LA-PAM was applied in excess of the suspended sediment available in the water column. The application rate should be reduced or the application postponed until higher loads of suspended sediment are available.</td>
</tr>
</tbody>
</table>

* The clear zone may not be noticeable if suspended sediment concentrations are extremely high.
Within this report, “overdosing” is defined as the use of an LA-PAM application rate higher than what the suspended sediment load can assimilate. This results in needlessly increasing the environmental exposure of LA-PAM without any additional seepage reduction benefit. An overdose of LA-PAM will result in the hydrated polymer remaining in the water column where it can travel downstream, as observed during many of the field applications having moderate and low SSC. Elevated LA-PAM concentrations observed within the fully developed clear zone at CAT-1 and HID-2 indicated an overdose of polymer for the given moderate (471 mg L$^{-1}$ at CAT-1) and low (31 mg L$^{-1}$ at HID-2) levels of suspended sediment. A second application, CAT-2, had nearly double the suspended sediment level of CAT-1 and resulted in only a partial development of the clear zone and no observable concentrations of LA-PAM below the treatment zone (Figure 9-6). Similar results were found with a second application at HID (HID-2; Figure 4-2). The consistency of back-to-back applications on the HID indicates that under the prevailing conditions, more than 31 mg L$^{-1}$ (HID-2) and less than 78 mg L$^{-1}$ (HID-1) of suspended sediment are needed to fully react with LA-PAM applied at a rate of 11.2 kg ha$^{-1}$ (10 lbs ca$^{-1}$).

The discrepancy in minimum suspended sediment levels for clear zone development at CAT and HID can be explained by three factors. First, the suspended sediment load (discharge x concentration) carried during HID-2 (0.5 kg s$^{-1}$) was only three times lower than the load carried by CAT-1 (0.5 kg s$^{-1}$). Although HID-2 only had seven percent of the suspended sediment concentration relative to CAT-1, it carried more than five times the water volume. For the formation of PAM-sediment flocs, the total weight (load) of sediment available is a more important indicator than the relative sediment concentration. Second, the faster water velocities, presence of several check structures, and more turbulent flow suggested a greater mixing at HID. A faster mixing rate increases the contact between the polymer and suspended sediment, increasing the potential for flocculation. Finally, the LA-PAM application rate at CAT-1 was 32-percent greater, requiring a greater load of suspended sediment to offset the additional polymer.

Overall, the partial development of the clear zone is the most ideal situation. In this scenario, flocculation has occurred and the remaining presence of suspended sediment in the water column suggests that the full mass of LA-PAM has been assimilated. In contrast, the full development of the clear zone indicated that all the suspended sediment has been removed and additional LA-PAM may remain in the water column.
9.2.3 Application Rate

The aggregated results suggest that an LA-PAM application rate of approximately 11.2 kg ha\(^{-1}\) (10 lbs ca\(^{-1}\)) overdosed small canals and may have been too low for larger canals. In the five smaller canals (SD, KC-1, MD, LAM-1, LAM-2), event-averaged time-weighted LA-PAM concentrations were higher, ranging from 1.5 to 4.0 mg L\(^{-1}\). In contrast, event-averaged time-weighted LA-PAM concentrations were lower in the seven larger canals (CAT-1, CAT-2, RFH-1, RFH-2, UNC, HID-1, HID-2), ranging from less than the detection limit of 0.1 mg L\(^{-1}\) to 0.8 mg L\(^{-1}\). These differences are the direct result of LA-PAM application rates being based on the wetted perimeter of the canal rather than water discharge or suspended sediment loads. For the canal reaches used in this study, canals were noted to carry more water per meter of wetted perimeter as they became larger (Figure 9-7). Small canals like SD and KC carried 0.06 m\(^3\) s\(^{-1}\) or less of water per meter of wetted perimeter compared to larger canals like UNC and HID, which carried an order of magnitude more water (0.8 m\(^3\) s\(^{-1}\)) per meter of wetted perimeter. Therefore, the base rate application of 11.2 kg ha\(^{-1}\) (10 lbs ca\(^{-1}\)) resulted in substantially higher dosages of LA-PAM to canals with lower water volumes than to canals with higher water volumes. For example, assuming an LA-PAM application rate of 8 g s\(^{-1}\), the concentration of LA-PAM in a fully-mixed canal ranged from 52 to 93 g m\(^{-3}\) at KC and SD to 0.4 to 0.5 g m\(^{-3}\) at UNC and HID, respectively.
Stated another way, LA-PAM concentrations would be expected to be lower in higher-volume canals due to dilution. Smaller-volume canals were found to be more likely overdosed with LA-PAM than larger canals (Figure 9-8).

It is possible that the base application rate of 11.2 kg ha\(^{-1}\) (10 lbs ca\(^{-1}\)) may be too low for larger-volume canals such as HID and UNC based on their ability to carry significantly more water volume per unit of wetted perimeter. For two experiments carried out in the largest volume canals, either PAM-sediment flocs were not observed to occur (UNC) or seepage rates were not observed to decline (HID-1). The lack of seepage reductions in HID-1 may have been due to the use of the stationary application method, however. Additional LA-PAM application studies need to be carried out in larger-volume canals under a greater variety of suspended sediment loads to verify these findings.

In summary, field-based results indicate that the basis of application rate on the wetted perimeter leads to the overdosing of smaller canals and a possible underdosing of larger canals. As the formation of PAM-sediment flocs is the primary mechanism for seepage reduction, it is more sensible to base the application rate of LA-PAM on the suspended sediment load carried within the canal prior to application. The use of a sediment-based application rate requires the knowledge of the canal flow rate and the concentration of suspended sediment prior to application. Only modest additional work would be necessary to develop sediment-based application guidelines, perhaps based on volume/volume sediment concentrations provided by an inexpensive and commonly available Imhoff cone rather than the volume/mass concentrations used here. If application guidelines based on wetted perimeter remain in use, then the addition of a sediment concentration multiplier should be adopted to reduce the overdosing of small canals and those with low SSC.

![Figure 9-7](image_url)  
Figure 9-7. Wetted perimeter compared against discharge of the canal reaches studied in this report.
Average LA-PAM concentrations relative to discharge and suspended sediment loading. Average LA-PAM concentrations included only the first 12 hours after the polymer was first added to the canal and were time-averaged, as the number of samples collected varied by site and time interval. Average maximum is the maximum LA-PAM concentration based on averages determined by hour.

9.2.4 Application Approach

If suspended sediment (Section 9.1.1) and water chemistry (Section 9.1.2) are sufficient to form PAM-sediment flocs, the last consideration is how to target the application of LA-PAM to the desired canal reach. Granular LA-PAM requires time to hydrate before it becomes reactive, so it will therefore travel downstream some distance before it reacts with suspended sediment and settles to the canal bottom. The guidelines for application of LA-PAM to water delivery canals (Susfalk et al., 2007) suggest that LA-PAM should be applied beyond the upper extent of the target reach the equivalent distance needed for water in the canal to travel 45 minutes. Additionally, the guidelines advocate that the start position be moved upstream the equivalent distance that it takes water in the canal to travel 10 minutes. These offsets were based on laboratory column studies that found LA-PAM to be reactive in as little as 10 minutes to as long as 45 minutes (Young et al., 2007b). The hydration laboratory study described above found complete LA-PAM hydration to occur in 28 to 29 minutes in 15 °C water.
However, in real-world water delivery canals, the clear zone was found to develop in as short as five minutes and as long as 208 minutes after the addition of LA-PAM to the water. The fastest hydration occurred on the Catlin Canal, ranging between five and six minutes during CAT-1 and CAT-2, respectively. At LAM-2, the clear zone required 20 minutes to begin development. For RFH-1 and RFH-2, the clear zone developed between 35 to 40 minutes after LA-PAM application. At these sites, where water temperatures ranged between 21.4 and 27.2 °C, no clear relationship was established between water temperature and the time needed to develop the clear zone. For example, a 4.8 °C difference was observed between water temperatures during CAT-1 and CAT-2, yet PAM-sediment flocs formed within six minutes at both sites. This indicates that other factors, such as mixing, water chemistry, and particle size, may play a more important role within this temperature range. The slowest development of the clear zone occurred in 132 minutes at MD (16 °C) and 208 minutes in SD (18.1 °C). Although MD had the coldest temperatures of all sites studied, it was hypothesized that the greater turbulence and water velocities caused the clear zone to develop faster than SD, which also had the lowest water velocity of all sites studied.

The range of time needed for the clear zone to develop indicates that the upstream offset distance used to provide time for LA-PAM to hydrate should be determined on a site-by-site basis. This is most easily accomplished by observing the time needed for the clear zone to develop at the downstream boundary of the target reach. Once the clear zone starts to develop, the user should multiply the elapsed time by the average water velocity to find the distance beyond the upstream boundary of the target reach where LA-PAM should be applied. For the canals studied, the calculated offset distances ranged from 196 m at CAT-1 to 2,149 m at SD.
10.0 SUMMARY

Granular LA-PAM was shown to reduce seepage rates from between 28 and 87 percent in 8 of 11 seepage rate experiments. Seepage reduction benefits remained effective throughout the remainder of the irrigation season at all four of the LARV sites where repeated measurements were conducted. Seepage reduction benefits were, however, not typically maintained through the winter when the canals were dewatered for several months, indicating the LA-PAM would have to be applied yearly. Water savings at the RFH and FL experiment sites was estimated to be $1.6 \times 10^6$ m$^3$ yr$^{-1}$ (1,300 acre ft$^{-1}$ yr$^{-1}$) and $8.5 \times 10^6$ m$^3$ yr$^{-1}$ (6,900 acre ft$^{-1}$ yr$^{-1}$) along each treated reach, respectively. If LA-PAM was applied to the entire length of each canal, water savings could be on the order of $6.7 \times 10^6$ m$^3$ yr$^{-1}$ (5,400 acre ft$^{-1}$ yr$^{-1}$) and $23.8 \times 10^6$ m$^3$ yr$^{-1}$ (19,300 acre ft$^{-1}$ yr$^{-1}$), respectively. On a basin scale, the application of LA-PAM to the entire length of the 14 major canals of the LARV could salvage approximately seven percent of the $1.043 \times 10^6$ m$^3$ (845,000 acre ft$^{-1}$) of water that these 14 canals divert annually. The above estimates are conservative, and assume a native seepage loss of 20 percent and that LA-PAM reduced seepage losses by 35 percent.

The use of granular LA-PAM has several benefits over traditional technologies. First is cost, as actual costs for individual applications ranged from $78$ km$^{-1}$ for smaller canals to $213$ km$^{-1}$ for larger canals. This represents a total annualized cost of between $111$ and $202$ ha yr$^{-1}$, or between 0.2 and 3 percent of the total annualized cost of conventional technologies, such as concrete or geomembranes. In addition to direct water savings, the lower cost of LA-PAM provides an opportunity to address local- and regional-scale problems that have previously been too cost prohibitive with traditional technologies. For example, a reduction in seepage from canals could be used to reduce the level of the groundwater table in areas characterized by already high groundwater levels, and it could be used to decrease the percolation of water through areas of naturally occurring soluble salts that have resulted in highly saline surface waters.

Second, the application of granular LA-PAM requires only modest personnel and equipment resources. Granular LA-PAM can be applied by stakeholders and/or canal employees that have been trained in the proper field safety and application protocols. Only modest application equipment is needed, ranging from hand seed spreaders to automated bucket seed spreaders attached to a moving boat or vehicle. However, current application devices tend to promote uneven and over-application of LA-PAM. The development of granular application devices that provide a temporally and spatially consistent application of LA-PAM at rates that will be used in the field is encouraged. Third, LA-PAM can easily be used to target seepage losses either in specific canal reaches and/or at specific times during the year. Targeting specific canal reaches known to have high seepage losses to minimize the release of LA-PAM into the environment is encouraged. The need to apply LA-PAM every year actually provides greater flexibility in controlling seepage rates compared to traditional methods. For example, some districts may choose to use LA-PAM only during drought conditions when water resources are scarce. Others might delay the use of LA-PAM by several weeks to provide canal seepage to adjacent resources (e.g., wetlands) during the early season when water is more plentiful. Yet others might delay LA-PAM application towards the middle or end of the irrigation season to help stretch the use of dwindling water resources. There is even the potential to halt seepage reduction benefits mid-season if they
are no longer required by disturbing the canal bottom and destroying the LA-PAM-induced seal.

It must be noted that LA-PAM was not always effective at reducing canal seepage, as evidenced by three of the 11 seepage rate experiments that were conducted. Several environmental conditions must be met to encourage the reaction and flocculation of LA-PAM with suspended sediment. Two of the most important factors include a sufficient level of suspended sediment in the water column and the presence of enough positively charged ions, particularly divalent cations, to promote cation bridging between LA-PAM and sediment. For the water delivery canals studied here, the presence of high enough suspended sediment levels was more critical, as nearly every canal had an acceptable cation concentration. Based on these limited number of trials and additional laboratory studies, it is suggested that LA-PAM only be added to canals having a SSC of more than 150 mg L\(^{-1}\) and a total dissolved concentration (TDS) of approximately greater than 200 mg L\(^{-1}\). Application of LA-PAM to canal water not meeting these thresholds is discouraged, as seepage rates will not likely be reduced and the LA-PAM can remain mobile in the water column and be transported downstream.

In addition to these thresholds, the rate of application and application technique should be managed to minimize the environmental exposure of and potential downstream impacts of LA-PAM, while maintaining its ability to reduce seepage rates. This can be primarily managed by controlling the rate at which LA-PAM is applied and by accounting for how environmental factors will affect the rate of flocculation formation. The field studies reported here showed that an application rate of 11 kg ha\(^{-1}\) (10 lbs ca\(^{-1}\)), based on the canal’s wetted perimeter, was sufficient to reduce seepage. Smaller-scale furrow (Susfalk, 2008) and laboratory (Young et al., 2007b) studies have suggested that LA-PAM can be effective at even lower application rates.

Elevated concentrations of LA-PAM observed in the water column downstream of the application reach indicated that an application rate of 11 kg ha\(^{-1}\) (10 lbs ca\(^{-1}\)) tended to over apply LA-PAM to smaller canals and possibly result in an under application to larger canals. These results indicate that the rate of LA-PAM addition should be related to the level of suspended sediment in the water column and not simply the wetted perimeter, as the mechanism for canal sealing depends upon the formation of PAM-sediment flocs, not the average wetted perimeter length. The concept of the Clear Zone Index was introduced as a diagnostic aid to help assess if the mass of LA-PAM being added is too great for the conditions. A partially developed clear zone is ideal as it indicates that flocculation has occurred and that little untreated LA-PAM remains in the water column available for transport downstream. Full development of the clear zone should be avoided because it indicates that LA-PAM has been added at a rate greater than can be assimilated by the concentration of suspended sediment in the water column.

The interaction of other environmental factors must also be taken into account to properly manage the application of LA-PAM. The interaction of water temperature and water velocity, for example, will determine how far LA-PAM will travel downstream before it reacts with suspended sediment. After it is added to the water, granular LA-PAM requires time to hydrate before it becomes reactive. Results suggest that the time needed for flocculation to occur was not entirely temperature dependent, indicating other site-specific factors such as water mixing, water chemistry, suspended sediment concentration and particle size may play a more important role. Slower hydration rates coupled with faster
water velocities, such as during early season snowmelt, result in LA-PAM traveling greater distances downstream before it becomes reactive. For LA-PAM to have reduced seepage at the upstream boundary of the target reach, the point of application had to have been between 196 m and 2,149 m upstream. In a worst-case scenario, LA-PAM could completely bypass the target reach and impact the canal further downstream.

The application of LA-PAM to unlined water delivery canals also carries several potential risks, including human health risks primarily from the residual AMD monomer, and environmental risks such as potential impacts on aquatic communities primarily from the polymer and potential groundwater contamination by AMD.

Elevated concentrations of LA-PAM downstream of the treated reach were not always observable. When they were, LA-PAM was typically detected between 4 to 9 hours depending on the length of time it took to apply LA-PAM. Peak concentrations, typically present on the order of tens of minutes, ranged from 4.8 mg L$^{-1}$ in larger canals to 16 mg L$^{-1}$ in smaller canals. These elevated concentrations were due to the application of LA-PAM at a rate in excess of the suspended sediment load it can react with in the canal. In addition, experiments having the highest peak concentrations and longest duration of elevated LA-PAM concentrations were observed during earlier experiments when the methodology was being developed. Excess polymer, represented by elevated LA-PAM concentrations, can remain in solution and travel significant distances downstream where it can be used by unsuspecting stakeholders, such as diversions to farms for use on crops, or consumption by livestock, for example.

There is also the potential for this excess LA-PAM to enter natural surface waters, such as receiving waters. The benthic macroinvertebrates (BMIs) studies presented here revealed that less tolerant BMI communities, such as found in natural systems, did have a detrimental response to LA-PAM. Some species, such as *Baetis tricaudatus* and *Simulium* sp., were found to be more sensitive to LA-PAM than others. In contrast, the aquatic community within water delivery canals is typically of low quality, reflective of the relatively harsh conditions due in part to activities like seasonal dewatering and the mechanical removal of riparian and in-stream vegetation to maintain the canal. The environmental consequences of using LA-PAM in water delivery canals were found to be relatively minor because the overall structure and mortality of these tolerant BMI communities did not appear to be affected by the application of LA-PAM, although a large number of organisms did move downstream during and immediately after application.

Three approaches can be utilized to minimize these potential application and transport risks. By far the most important approach is to prevent the addition of LA-PAM in excess of the suspended sediment with which it can react. If the mass of LA-PAM added to the canal is consumed within the treatment reach, it will not be available for downstream transport. It is suggested that LA-PAM be applied only under turbid conditions and techniques such as the Clear Zone Index used to assure that the load of suspended sediment is much greater than the added polymer. It is strongly advised that current application guidelines based on a canal’s wetted perimeter be depreciated in favor of the development of application rates based on suspended sediment concentration or load. The second approach to minimize downstream risks is to decrease the potential for LA-PAM-treated water to enter farms by closure of downstream offtake gates during and for a period after LA-PAM application. The last approach is to not apply LA-PAM in circumstances where it could enter natural surface...
waters, as the release of polyacrylamide into receiving and other natural surface waters must be avoided.

The most likely human health risk is to the persons applying LA-PAM, either through inhalation or through accidents resulting from slipping on the slick polymer as it hydrates. These risks are mitigated through the proper use of safety guidelines presented in Susfalk et al. (2007). The incidental release of the residual AMD found in LA-PAM also presents environmental and/or human health risk if it were to accumulate in groundwater. The AMD monomer is a cumulative neurotoxin and a suspected human carcinogen, whose risks related to application in water delivery canals were more fully discussed in Young et al. (2007a).

During the field applications presented here, the concentrations of AMD measured in the canal water were orders of magnitude below the chronic levels needed to impact human health. Over 50 percent of the 36 samples analyzed were below the detection limit of $0.1 \mu g L^{-1}$, while only three exceeded the U.S. EPA limit of $0.5 \mu g L^{-1}$ (by less than 30%). There is evidence that elevated AMD concentrations can be associated with elevated LA-PAM concentrations, so it might be expected that the precautions taken to minimize LA-PAM concentrations discussed above can also minimize AMD concentrations. Once in the water, the AMD monomer is mobile and can enter groundwater, although at a reduced rate, as PAM-sediment flocculation starts to reduce seepage. Greater mixing with bulk canal water and dilution with groundwater result in AMD concentrations becoming lower. At one site, surrogate AMD concentrations entering groundwater from the canal were diluted by over four orders of magnitude. Furthermore, microbial degradation will decrease AMD concentrations even further, with an estimated half-life of 30 to 42 hours in natural systems. Transport models indicated that an AMD concentration 10 times greater than the maximum concentration actually observed in canal water would be undetectable within 25 m of the canal due to microbial degradation and dilution processes. Therefore, the contamination of groundwater by AMD associated with the application of LA-PAM to water delivery canals using the methods of Susfalk et al. (2007) was considered to be very unlikely.

In conclusion, the application of LA-PAM to water delivery canals can substantially reduce seepage losses. However, proper field conditions (e.g., sufficient dissolved solids and suspended sediment) must be present for LA-PAM to flocculate out the suspended sediment that subsequently causes seepage reduction. LA-PAM should only be applied where acceptable conditions exist, and the mass of LA-PAM added should not be greater than the assimilative capacity of suspended sediment. This latter condition cannot be accounted for by using an application rate based simply on wetted perimeter, as is commonly done. Failure to account for these conditions increases the likelihood that LA-PAM will be overapplied, resulting in the downstream transport of LA-PAM and AMD. Human health risk, primarily associated with contact during application, can be minimized with education and the use of safe handling techniques. Contamination of groundwater by AMD was considered to be very unlikely due to dilution and microbial degradation, assuming proper application guidelines were followed. However, as a conservative measure and lacking additional information, it is suggested that the application of LA-PAM in situations where it could enter natural water systems due to potential impacts of the polymer on aquatic systems be prohibited. Prior to any individual application, stakeholders and/or agencies responsible for the application must assess if the potential, site-specific risks outweigh the benefits of using LA-PAM for seepage control.
11.0 PROPOSED RECOMMENDATIONS

11.1 Application Methods and Techniques

The current practice of utilizing LA-PAM application rates based on wetted perimeter must be replaced with application rates based on the load of suspended sediment in the canal when the polymer is applied.

An initial application rate based on sediment load could be developed using the data presented here. However, additional studies will be needed to refine the use of suspended sediment load-based application rates under a variety of field conditions.

The clear zone concept should be developed into a field application protocol that could help determine proper application rates under field conditions either immediately prior to or during the initial application of LA-PAM.

Physical methods used to apply granular LA-PAM could be improved to significantly further reduce worker contact with LA-PAM and to promote a more consistent application both across and down the canal.

There is a tendency by end-users to over-apply LA-PAM, due to its low relative weight, to promote flocculation in canals. The development of alternatives should attempt to address this issue. One possible approach would be the development of application devices to specifically limit the amount of polymer added based on environmental conditions. A second approach would be the development of an LA-PAM-based product containing a lower percentage of active ingredient through the addition of an environmentally safe filler material.

Training of agency personnel and end-users must be updated as application procedures are modified and new techniques are developed.

A better understanding is needed regarding when LA-PAM becomes effective. This could be accomplished through additional research on the rate and factors that affect PAM-sediment floc formation. This information would improve the application protocols and provide better estimates of potential downstream movement of LA-PAM.

11.2 Seepage Reduction and Benefits

Several of the minimum field conditions necessary for an effective application of LA-PAM were delineated. However, little is known about how application techniques and field conditions interact to affect seepage reduction benefits. Additional studies will need to be conducted to better understand the mechanisms involved in the sealing processes and how the application and field conditions can be manipulated to produce the most effective seal. This improved understanding could help to answer questions such as:

- How important is in-canal mixing to the flocculation and sealing processes that ultimately reduce seepage?
- Does a greater application rate of LA-PAM or application during a higher suspended sediment load accelerate and/or increase the strength of the seal?
- What are the impacts of environmental conditions, such as water temperature and velocity, on the strength of the seal formed?
• How do site-specific characteristics, such as suspended sediment particle composition and size, affect flocculation and sealing?

Results presented here indicate that seepage reduction benefits primarily last only during the season they were applied, at least for canals that are seasonally de-watered. Empirical observations suggest that seepage reductions may last multiple years in some canal systems. Additional effort is needed to obtain an in-depth understanding of the factors that contribute to the longevity, and how these factors could be modified to optimize the longevity of a sealed canal.

The savings of water resulting from LA-PAM application needs to be determined under a wider variety of conditions. This requires a better knowledge of seepage losses under native conditions as well as an improved understanding of the factors that control both the temporal and spatial variation typically observed in seepage losses. This information will help to quantify the mechanisms and extent of the underlying issue, seepage loss, as well as to provide improved estimates of the net benefits of LA-PAM application.

The potential benefits and costs associated with LA-PAM application need to be quantified in a greater number of local and regional settings. The cost and benefits of management approaches utilizing LA-PAM, such as extending the irrigation season or improving the ability to move and control water in a canal system, need to be more fully quantified.

Seepage reduction benefits of other PAM forms (e.g. dissolved, emulsion) need to be studied and compared against the use of granular forms.

11.3 Risks and Alternatives

Results reported here and elsewhere suggest that LA-PAM in water delivery canals does not pose a large risk if used properly. However, there is still a need for a greater understanding of the potential impacts that LA-PAM, AMD, and their degradation products have on the environment, including long-term bioaccumulation rates.

Additional studies must be conducted to examine the potential impacts that LA-PAM has on native aquatic communities, as the accidental or unintentional release of LA-PAM into natural streams and rivers remains a concern. Research presented here only examined LA-PAM as a proximate factor affecting benthic macroinvertebrate indicators. Until future studies are carried out that examine the ultimate factors impacting native aquatic communities, LA-PAM must not be applied to canals segments whose treated canal water returns to native streams and rivers.

Future efforts regarding the use of a flocculent-based canal sealant technology should progress down three parallel pathways. The first pathway involves the reduction in the amount of LA-PAM and AMD released into the environment through the optimization of the canal sealing process as recommended above. This requires, in addition, that significant resources be utilized for the education of stakeholders and end users in the proper use of LA-PAM that minimizes environmental release. Primary tenets of any education campaign must include: 1) abandoning the use of LA-PAM if environmental conditions are not suitable, or if potential downstream risks are unacceptable; 2) limiting application to specific canal reaches known to have higher seepage rates; 3) limiting the application of LA-PAM to a rate that
does not exceed the assimilation capacity of suspended sediment in the canal water, and; 4) use of proper safety equipment.

The second pathway of future research is the development of “greener” PAM products that substantially reduce or eliminate the residual AMD content found in current formulations. In addition to eliminating the perceived risks currently posed by incidental AMD release, a “greener” formulation would have nearly the same chemical and physical properties as today’s formulations and therefore be a direct replacement. Therefore, existing knowledge, resources, and methods pertaining to LA-PAM will be directly applicable to a “greener” formulation.

The final pathway is the development of an alternative to LA-PAM. For an alternative to be adopted, it must share at least two qualities with LA-PAM: be relatively inexpensive and effectively reduce seepage when used in low concentrations of 1 ppm or less. Although this latter requirement appears to make the discovery of an effective alternative non-trivial, resources should continue to be devoted to finding a suitable alternative. An alternative will likely have substantially different chemical and physical properties than LA-PAM, requiring additional resources necessary to conduct basic studies to determine the benefits, risks, and application methods of the new alternative.

12.0 REFERENCES


SonTek/YSI., 2003. Flowtracker handheld ADV principles of operation, SonTek/YSI, San Diego, CA.


U.S. Food and Drug Administration, Department of Health and Human Services, 2006. Food Additives Permitted in Feed and Drinking Water of Animals. 21CFR573.120. Washington, DC.


Groundwater monitoring wells were installed at locations along the Catlin Canal that displayed evidence of a high groundwater table. Such evidence includes dense vegetation, wetland vegetation (such as cattails), and standing water above the ground surface. The wells were drilled by a Giddings rig mounted to a truck bed. The wells were drilled to a great sufficient depth below the ground surface so that the groundwater table would be located at an elevation above the bottom of the well. This enabled the depth of the water column in each well to be measured.

Each well was composed of 6.35-cm diameter (2.5-inch-diameter) PVC casing and contained 0.4-mm-wide (0.016-inch-wide) horizontal perforations about every centimeter along the length of the pipe. A plug was placed in the bottom of the casing. To keep precipitation from directly entering the monitoring wells, a cap was placed on the top of well (typically located a couple feet above the ground surface) and bentonite (a low-permeability clay) was packed between the outside of the well and the ground surface. Small holes were drilled in the well caps so that the air pressure inside of the monitoring wells remained at atmospheric pressure. These holes were drilled at an upward angle so that precipitation could not enter the inside of the well.

A pressure transducer was installed near the bottom of each well at an elevation below the groundwater surface. Each pressure transducer measured absolute pressure (atmospheric pressure plus gage pressure) below the groundwater surface every 30 minutes. To account for atmospheric pressure variability, an atmospheric pressure transducer was installed at a midpoint location along the study reach. The pressure transducers installed in the wells recorded absolute pressure. To calculate the height of the water column above the pressure transducers in the wells, gage pressure was used along with mathematical conversions involving the unit weight of water. The gage pressure was calculated as the difference between the absolute pressure (recorded by the pressure transducers in the monitoring wells) and atmospheric pressure (collected by the atmospheric pressure transducer near the center of the study reach). Manual measurements of the groundwater elevation in each well were collected periodically throughout the 2007 water year to ensure the pressure transducers were collecting approximately accurate stage measurements.

A total of five batteries of groundwater monitoring wells were installed along the Catlin Canal, each approximately 0.5 miles apart. The spacing of these batteries was affected by the acquisition of permission for installation from land owners and safe access for drilling with a Giddings rig mounted to a truck. Three batteries were primarily studied (batteries 1, 4, and 5), since no pressure transducers were available for battery 2 and the wells composing battery 3 could not be drilled to a sufficient depth suitable for consistently measuring water depths. Pressure transducers were installed in each monitoring well within the three batteries on which this study focused. Each battery contained three monitoring wells located on the downgradient side of the canal (on the north side of the canal) extending in a straight line perpendicular to the canal. The wells closest to the canal and the middle wells were typically spaced approximately 10 m (32.8 ft) apart, and the middle wells and wells furthest from the canal were typically spaced approximately 15 m (49.2 ft) apart. The wells closest to the canal were located within a couple of meters of the canal road on downgradient side. In other words, the canal road was located between the canal and the closest monitoring well for each battery. Some batteries included a monitoring well on the upgradient side of the canal to
observe the groundwater elevation relative to the canal water surface elevation and the affects of irrigation on the groundwater table on the upgradient side of the canal. However, not every battery included a monitoring well on the upgradient side of the canal. Upgradient wells were not installed if access was difficult or unsafe for drilling, if drilling was prohibited by land owners, or if the groundwater table was located at a depth so great below the ground surface that it could not be reached using the available drilling equipment. A pressure transducer was installed within the canal in line with the monitoring wells of each battery to measure the canal water surface level relative to the groundwater levels. The pressure transducer measuring atmospheric pressure was located in close proximity to battery 4. A map of the Catlin Canal is shown in Figure A-1 and displays the locations of the atmospheric pressure transducer and the monitoring wells of the three batteries studied.

![Figure A-1. Map of the monitoring wells studied on the Catlin Canal.](image)

The gradients of the groundwater table between the canal water surface and each monitoring well were estimated. The lateral distances separating the canal and the wells and the elevation differences between the pressure transducers in the canal and the monitoring wells were measured using survey equipment. From the survey data, relative elevations of the canal water surface and the groundwater surface inside each monitoring well were calculated. The average groundwater gradients before and after the 2007 PAM application for well battery 1, 4, and 5 appear in Tables A-1, A-2, and A-3, respectively.
### Table A-1. Average groundwater gradients for Catlin Canal well battery 1.

<table>
<thead>
<tr>
<th>Time Period</th>
<th>Average Groundwater Gradient Over the Time Period</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Canal to Well 1A</td>
</tr>
<tr>
<td>- 2 Months</td>
<td>0.260</td>
</tr>
<tr>
<td>- 1 Month</td>
<td>0.275</td>
</tr>
<tr>
<td>- 2 Weeks</td>
<td>0.277</td>
</tr>
<tr>
<td>- 1 Week</td>
<td>0.271</td>
</tr>
<tr>
<td>+ 1 Week</td>
<td>0.292</td>
</tr>
<tr>
<td>+ 2 Weeks</td>
<td>0.294</td>
</tr>
<tr>
<td>+ 1 Month</td>
<td>0.312</td>
</tr>
<tr>
<td>+ 2 Months</td>
<td>0.338</td>
</tr>
</tbody>
</table>

### Table A-2. Average groundwater gradients for Catlin Canal well battery 4.

<table>
<thead>
<tr>
<th>Time Period</th>
<th>Average Groundwater Gradient Over the Time Period</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Canal to Well 4A</td>
</tr>
<tr>
<td>- 2 Months</td>
<td>0.567</td>
</tr>
<tr>
<td>- 1 Month</td>
<td>0.570</td>
</tr>
<tr>
<td>- 2 Weeks</td>
<td>0.571</td>
</tr>
<tr>
<td>- 1 Week</td>
<td>0.573</td>
</tr>
<tr>
<td>+ 1 Week</td>
<td>0.580</td>
</tr>
<tr>
<td>+ 2 Weeks</td>
<td>0.579</td>
</tr>
<tr>
<td>+ 1 Month</td>
<td>0.585</td>
</tr>
<tr>
<td>+ 2 Months</td>
<td>0.583</td>
</tr>
</tbody>
</table>

### Table A-3. Average groundwater gradients for Catlin Canal well battery 5.

<table>
<thead>
<tr>
<th>Time Period</th>
<th>Average Groundwater Gradient Over the Time Period</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Canal to Well 5A</td>
</tr>
<tr>
<td>- 2 Months</td>
<td>0.327</td>
</tr>
<tr>
<td>- 1 Month</td>
<td>0.341</td>
</tr>
<tr>
<td>- 2 Weeks</td>
<td>0.328</td>
</tr>
<tr>
<td>- 1 Week</td>
<td>0.313</td>
</tr>
<tr>
<td>+ 1 Week</td>
<td>0.337</td>
</tr>
<tr>
<td>+ 2 Weeks</td>
<td>0.326</td>
</tr>
<tr>
<td>+ 1 Month</td>
<td>0.332</td>
</tr>
<tr>
<td>+ 2 Months</td>
<td>0.333</td>
</tr>
</tbody>
</table>
In each of the batteries, the average groundwater gradient between the canal and the closest monitoring well (well A) increased over a period of one week after the PAM application. The canal water levels remained fairly stable, which means that the groundwater level in the well closest to the canal likely decreased in elevation relative to the canal water surface. A reduction in seepage is supported by this observation since the amount of water traveling through the channel perimeter into the wells would decrease, making the gradient increase.

The gradients between wells A and B, for each battery, appear to have decreased on average over the first week after PAM application. This also supports the observation that the water levels in the wells closest to the canal were dropping as a result of seepage reduction, since the difference between the water levels in well A and well B became smaller.

The effects of the PAM application on the gradient between wells B and C (furthest from the canal), for each battery, were not as evident. The gradients appear to drop by a small amount one to two months after PAM application. This could be a result of the longer travel time required for the groundwater table to drop further from the canal as a result of seepage reduction. However, it could be attributed to the decreased canal flow rates and canal water surface levels that occur later in the water year.