Water Quality Appendix

Platte River Endangered Species RIP FEIS
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Contents

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Introduction

The purpose of this technical report is to document the data and analysis that were undertaken in developing the sections included in the EIS for the Platte River Recovery Program. During the development of the EIS, a number of water quality reports w ere generated. Essentially each time that a new set of alternatives were developed, a new report was also generated. This report will integrate the sections of those early reports that are still current with the most recent impact analysis.

For purposes of the water quality impacts analysis, the potential effects of the Program extend from the North Platte River at Seminoe Reservoir in Wyoming and the South Platte River near Julesburg, Colorado, to near Grand Island, Nebraska. The EIS uses indicator variables to evaluate the effects of the Program. This report will document the process that was followed in evaluating potential water quality indicators. This report will then track through the impact analysis by river reach by applying the indicators.

Much of the water quality impact assessment for the Program is based on existing data. Most of the data have been collected by the U.S. Geological Survey and were downloaded from their National Water Information System (NWIS). Where the NWIS data were either not adequate or indicated a potential water quality problem, the Program collected additional samples for analysis to better define conditions. Much of the analysis of these historic data that were used to develop impact evaluation relationships could not be included in the EIS due to space limitations, but will be presented in this report.

The report is organized by river reach or affected area, beginning with the North Platte River in Wyoming, picking up the South Platte River in Colorado, then proceeding downstream on the Central Platte River in Nebraska. Each section is begins with an affected environment section, which evaluates and screens the available data. The next section is an environmental consequences section that begins with a method of analysis section. The methods of analysis section follows up the analysis in the affected environment section by screening indicators and developing the method to be used to compare the various alternatives with existing conditions. The existing conditions are mostly developed for each indicator from the present conditions as defined in model output based on the method used to develop the indicator data for comparing alternatives. The same method is then used to develop the indicator data for the alternatives. The hydrologic model data from the Program's North Platte, South Platte, and Central Platte models are used in the alternatives comparisons in most cases. An exception is the section on North Platte tributaries in Nebraska that will only be indirectly affected by one alternative. The analyses are based on monthly or daily model flow output depending on the available historic data. For sediment related effects, the output from the sediment-vegetation model were used. There is also a section on the effects of the alternatives on Clean Water Act Section 303(d) impaired waters . The last sections of the appendix evaluates unpublished data on contaminants in fish and bird eggs.

North Platte River Basin, Wyoming

Affected environment

The mainstem of the North Platte River, including mainstem reservoirs, are included in the potentially affected environment in Wyoming. This section will provide a description of background water quality in the mainstem of the North Platte, including Seminoe, Pathfinder, Alcova, and Glendo reservoirs, and the intervening river reaches. The basin, including the location of the reservoirs, is shown on Figure 1. Guernsey Reservoir is also located on the mainstem of the North Platte River, but it is essentially dry during the nonirrigation season and would not be greatly affected.

Selenium is the only water quality problem in the North Platte River as defined by Clean Water Act compliance (see above). The sources of selenium and its trends in the mainstem of the river were described above. Some of the interrelationships of selenium with other water quality constituents will be explored in this section.

The various Program alternatives will change the operation of the reservoirs in Reclamation's North Platte system. These changes in operations have the potential to affect the temperature and dissolved oxygen (DO) regimes of the reservoirs with consequent affects on coldwater fisheries in the reservoirs and in the river reaches downstream. There are important riverine fisheries downstream from Seminoe, Pathfinder, and Alcova reservoirs that could be affected. The temperature and DO of these river reaches and the reservoirs will be described and the possible effects of the alternatives on temperature and DO will be evaluated.

Existing Water Quality – Total Dissolved Solids

As is typical of rivers in the semi-arid American West, the total dissolved solids (TDS) in the North Platte River shows a general increase with distance downstream. For this reason, TDS (and its more easily measured surrogate, specific electrical conductance [EC]), are major concerns in Western rivers. Since there are more EC measurements than TDS analyses at most of the sites, EC will be used to characterize the TDS of the North Platte River. The relationship between TDS and EC will be described in more detail in a later section, but in general, the TDS concentration in mg/L is about ⅔ of the EC value in μS/cm.

Figure 2 shows the EC of the North Platte River at 15 sites. Although the periods of record for the sites vary in both length and the years in which they were in existence, the medians on Figure 2 show a relatively smooth and gradual increase from above Pathfinder Reservoir to the State Line. There is a gradual increase in EC between the gage above Pathfinder Reservoir to the gage at Goose Egg. The EC then remains constant between the Goose Egg gage and the one above Poison Spider Creek. Downstream from Poison Spider Creek, the median EC of the river shows a steady increase to the gage below Casper. From the gage below Casper until the one at Lingle, the median EC of the river fluctuates slightly, but remains generally constant. There follows a final increase in the median EC of the river between the Lingle and State Line gages (Figure 2).

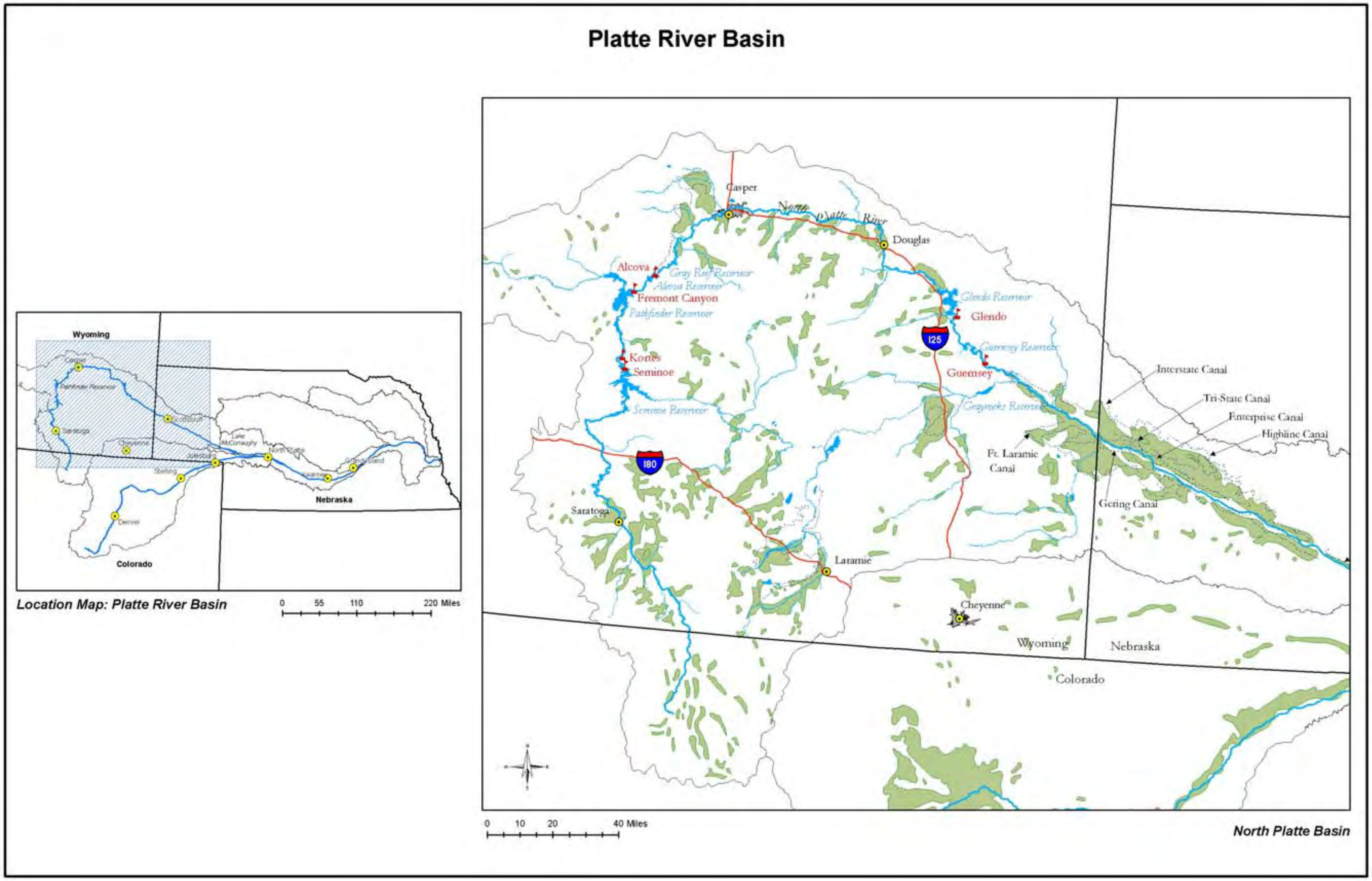

Platte River Basin
Wyoming
Colorado
Nebraska
Location Map: Platte River Basin
0 55 110 220 Miles
Casper
North Platte River
Douglas
Alcova
Gray Reef Reservoir
Alcova Reservoir
Fremont Canyon
Pathfinder Reservoir
Kortes
Seminoe
Seminoe Reservoir
Glendo Reservoir
Glendo
I25
Guernsey Reservoir
Guernsey
Grayrocks Reservoir
Interstate Canal
Tri-State Canal
Enterprise Canal
Highline Canal
Ft. Laramie Canal
Gering Canal
I80
Saratoga
Laramie
Cheyenne
Wyoming
Nebraska
Colorado
0 10 20 40 Miles
North Platte Basin

Figure 1

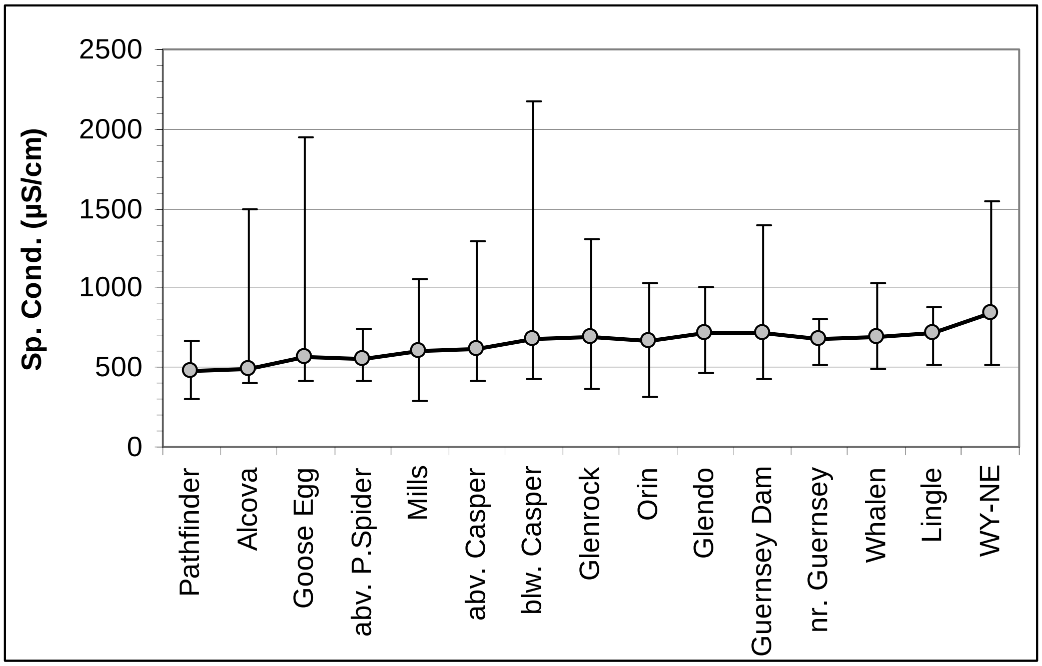


Figure 2. Median EC of the North Platte River between Seminoe Dam and the State Line
(Error bars represent the minimum and maximum values)

Although the median EC shows a relatively smooth distribution over the length of the North Platte River from above Pathfinder Reservoir to the State Line, the maximum EC values show a large amount of variation among the sites (Figure 2). The minimum EC values do not show anywhere near the amount of variation that the maximum values show. The much larger difference between the maximum and median EC relative to the difference between the minimum and median EC indicates that the data are skewed high. For this reason, much of the following analysis is based on log-transformed data. A log-transformation is the usual way of normalizing water quality data such as those shown on Figure 2.

There are a number of highly saline tributaries between Alcova Dam and Mills, *i.e.* have a high TDS (and EC). Lone Tree Gulch and Bates and Poison Spring creeks enter the river between the Alcova and Goose Egg gages. There are no significant inflows between the gage near Goose Egg and the one above Poison Spider Creek, but a number of tributaries enter between that gage and the one at Mills. The tributaries and most of the mainstem gages on Figure 2 are shown on Figure 3, which shows median and maximum EC values for each of the sites.

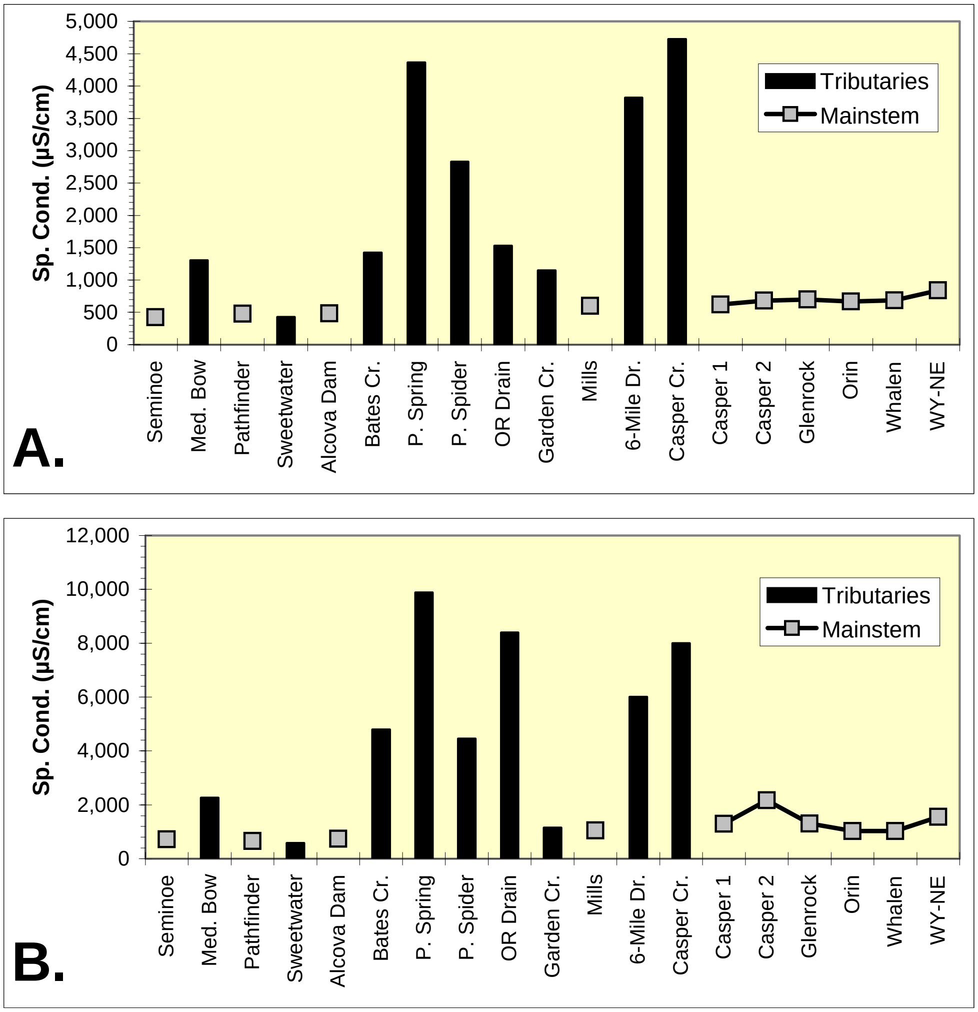


Figure 3. Specific conductance of the mainstem (A) of the North Platte River and nine of its tributaries (B) from upstream from Seminoe Reservoir to the Wyoming-Nebraska State line

The median EC plot on Figure 3A illustrates the gradual increase in the EC of the North Platte River from above Seminoe Reservoir to the State Line. The largest increases in the median EC are between Alcova Dam and Mills (486 to 600 µS/cm) and between Whalen Diversion Dam and the State Line (685 to 840 µS/cm). The tributaries plotted on figure 3A relative to the point at which they enter the river. Although the tributaries have a very high median EC, they carry comparatively little flow. Because the flow in the river is so much greater than the tributaries, there is significant dilution, but not enough to prevent some increase in its EC.

As was the case with selenium, the maximum EC of the tributaries (Figure 3B) is very much higher than the median EC (compare the *y*-axes of figures 1A and 1B). Unlike the median ECs in the mainstem, the maxima do not show the gradual increase with distance downstream. There is a particularly dramatic peak at the site labeled Casper 2, which is located downstream from Casper (Casper 1 is upstream). Casper Creek, which is highly saline with a median EC of near 5,000 µS/cm, enters above the upstream Casper gage (Figure 3) and could not be a factor in the maximum EC at the gage below Casper.

Some of the differences in the maximum EC are likely a reflection of the differences in the periods for which data are available at the various gages. Those periods differ considerably in some cases and in some cases there is no overlap. This is described below.

Relationships between flow, EC and TDS

In addition to the EC data summarized above, TDS data for the mainstem gages on the North Platte River were retrieved from the U.S. Geological Survey's National Water Information System (NWIS). The periods of record for each of the gages for which there are flow, EC, and TDS data available on the North Platte mainstem in Nebraska are summarized in Table 1. The peak in EC noted above for the gage downstream from Casper was noted above. As is shown in Table 1, the period of record for that gage does not coincide with those of any of the other gages – its period of record ended before those of any of the other gages, except for two (near Goose Egg and below Guernsey), began. For this reason, the maximum EC shown above at the gage below Casper could have originated in any of the tributaries that enter between that gage and the one near Goose Egg. Flow measurements are currently made at most of the gages, but the end date reflects when water quality measurements ended at the sites. Where possible, TDS data were supplemented by using a regression between TDS and EC. At some of the gages shown in Table 1, there were no TDS data and these gages could not be used for some of the water quality analysis. For example, there were no TDS data for the gages below Guernsey Dam and Whalen Diversion Dam.

Table 1. Periods of record of flow, specific conductance, and TDS at selected sites in the North Platte Basin in Wyoming

Gage	Begin Date	End Date
North Platte River above Seminoe Reservoir	12/4/1960	9/14/2000
Medicine Bow River above Seminoe Reservoir	7/28/1965	6/29/1993
North Platte River above Pathfinder Reservoir	8/25/1969	7/31/1989
Sweetwater River near Alcova	10/1/1964	5/25/1990
North Platte River at Alcova Dam	10/1/1965	9/30/1998
North Platte River near Goose Egg	10/1/1957	10/21/198
North Platte River above Poison Spider Cr.	11/9/1977	9/6/1995
North Platte River at Casper	10/5/1971	9/7/1994
North Platte River below Casper	6/1/1949	9/30/1959
North Platte River near Glenrock	12/8/1960	4/24/1986
North Platte River at Orin	7/14/1966	9/15/2004
North Platte River below Glendo Reservoir	10/13/1966	9/29/1988
North Platte River below Guernsey Reservoir	12/7/1950	5/6/1986
North Platte River below Whalen Diversion Dam	7/17/1970	10/8/1976
North Platte River near Lingle	7/18/1969	6/10/1975
North Platte River at WY-NE State Line	8/19/1964	9/13/2004

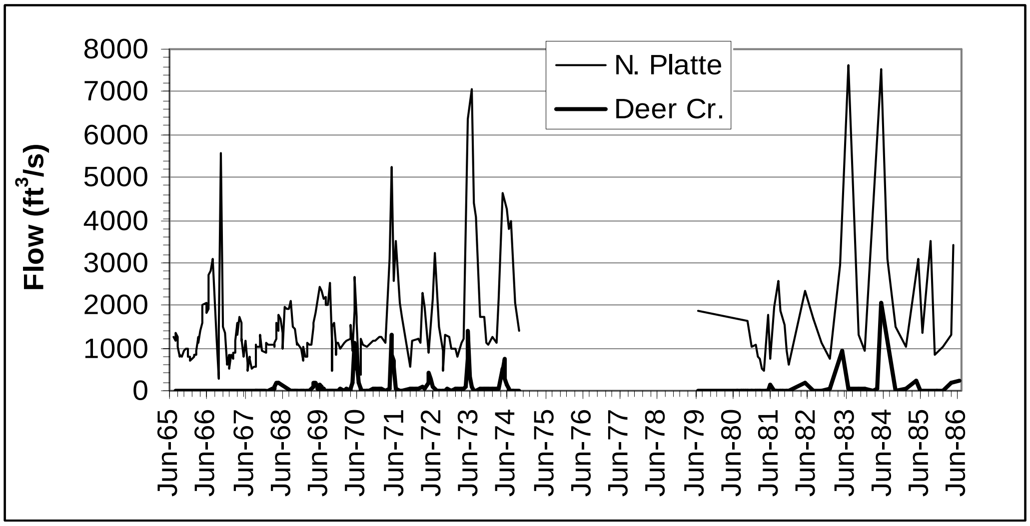
In rivers where there is an inflow of high TDS water from tributaries or base flow (ground water), there tend to be good relationships between TDS (and EC) and flow. The relationship should be inverse and reflect the greater dilution of these inflows at higher river flows. The relationships between TDS and flow are shown in Table 2, along with the relationships between TDS and EC. All of the regressions between TDS and flow are based on log-transformed data.

All of the TDS-flow regressions are statistically significant, except for the one above Pathfinder, where there were only 12 observations. The r^2-values for 7 of the 10 TDS-flow regressions are around 0.7" (Table 2). The regressions with lower r^2-values include those for the North Platte above Pathfinder, at Alcova, and near Glenrock. The Pathfinder site is below Seminoe Dam and

Table 2. Regressions of TDS on flow and specific conductance (EC) in the North Platte River, Wyoming							
Independent Variable	Site	Equation		r^2	F	Prob. > F
		Intercept	Slope			
Flow: $TDS = ax^b$	N. Platte above Seminoe	1384.84	-0.25768	0.689	736.4	2.26E-86
	Medicine Bow River	2568.52	-0.25289	0.735	661.5	8.69E-71
	North Platte above Pathfinder	301.453	0.00092	0.000	0.0002	0.988224
	Sweetwater River	625.64	-0.20997	0.686	425.3	6.90E-51
	North Platte at Alcova Dam	480.49	-0.06075	0.147	44.3	1.67E-10
	North Platte below Casper	4157.33	-0.31323	0.766	1,124.4	1.9E-110
	North Platte near Glenrock	3002.42	-0.26262	0.423	227.2	6.70E-39
	North Platte at Orin	4820.06	-0.32728	0.674	516.9	8.72E-63
	North Platte River near Lingle	1024.92	-0.12227	0.715	216.1	3.49E-25
	North Platte River at WY-NE State	1469.22	-0.15876	0.707	584.7	1.68E-66
Sp. Cond.: $TDS = a + bx$	N. Platte above Seminoe	-6.80	0.65086	0.937	5,659.9	2.5E-230
	Medicine Bow River	-71.55	0.77995	0.962	2,865.7	1.25E-82
	North Platte above Pathfinder	53.27	0.49248	0.800	40.1	8.51E-05
	Sweetwater River	13.08	0.60109	0.884	1,169.1	1.54E-73
	North Platte at Alcova Dam	89.048	0.44795	0.586	294.1	1.14E-41
	North Platte below Casper	-61.983	0.72861	0.933	4,771.4	1.02E-203
	North Platte near Glenrock	-51.982	0.73835	0.966	8,934.2	1.30E-230
	North Platte at Orin	-28.462	0.69304	0.902	1,395.7	3.52E-78
	North Platte River near Lingle	-56.287	0.73383	0.975	2,929.2	1.89E-62
	North Platte River at WY-NE State	-4.963	0.66823	0.854	1416.6	4E-103

the Alcova site is below Alcova Dam, as the gage name states. As was noted above, the TDS-flow relationship is usually a reflection of the dilution of a relatively steady inflow of higher TDS water (most often base flow, but occasionally from a continuously discharging point source) by a relatively large upstream source of low TDS runoff. At sites below dams, the dilution will have occurred upstream within the reservoir proper and over the course of weeks or even months, rather than instantaneously, as in a stream. Actually, it is somewhat surprising that there is a significant relationship between TDS and flow below Alcova Dam.

The reason for the reduced r^2 for the regression near Glenrock is unclear, but the usual reason is the variable influence of water of different quality from different areas within the drainage basin. The majority of the flow at Glenrock would be from the North Platte River mainstem. Based on gage records, Deer Creek is the only significant tributary between the Casper and Glenrock gages. Time series plots of the flows in the North Platte River near Glenrock and Deer Creek for the period of record of the Deer Creek gage are shown on Figure 4. A table summarizing the flow data at the 2 gages is also included on Figure 4. Although Deer Creek may be completely dry at times, at other times the flow can be a significant portion of the flow of the North Platte River at the Glenrock gage. Each of the peak flows in Deer Creek coincides with a large peak flow in the North Platte River. To illustrate how Deer Creek may affect the flow in the North Platte River, the peak daily flow at the Deer Creek gage during the period that water quality data were being collected was 3920 ft^3/s. The flow in the river was equal to that flow plus the previous days flow in the river, indicating that the majority of the flow in the river was from Deer Creek. The contributions from Deer Creek under these circumstances would affect the TDS-flow relationship at the river gage. Under most conditions, the TDS-flow relationship



North Platte River	Minimum	303	Median	1180	Maximum	7620
Deer Creek	Minimum	0	Median	24	Maximum	2050
		0 flow 2 % of the time				

Figure 4. Flows in the North Platte River near Glenrock and Deer Creek below Millar Wasteway for the period of record of the Deer Creek gage

should be dominated by conditions in the mainstem of the North Platte River upstream, but the periodic influence of Deer Creek could make the relationship inconsistent and reduce the r^2 of the regression relationship between TDS and flow.

The summary table at the bottom of Figure 4 indicates that Deer Creek carries no flow at times. There were 3 days on which there were no flows in Deer Creek. These are not to be confused with the gap in the record for most of the late 1970s. The Deer Creek flows on most dates are difficult to see on Figure 4, and would show much better on a log-scale plot. However, the 0 flows prevent the log-scale from being used. The plotted flows are those associated with water quality data and are not gage records. For the more recent water quality data at both sites, the flows are instantaneous measurements, but the earlier data have associated mean daily flows.

Figure 5 shows the relationship between TDS and flow at the Deer Creek gage. Figure 5A shows the log-transformed regression that is equivalent to those of the North Platte River shown in Table 2. The slope of the regression is between 0.2 and 0.3, like the majority of those of the North Platte gages in Table 2. The r^2 of the regression at 0.8 is better than any of those of the North Platte sites in Table 2. However, the plot of the data indicates that the log-linear regression may not be the best fit to the data. A comparison of the data to the straight line on Figure 5A indicates that the TDS at both high and low flow plot below the line, while the TDS at intermediate flows plot above the line. Even on a log-log plot, the data form a sickle or curved shape. Where the regression on Figure 5A is a log-linear power fit, the regression on Figure 5B is a nonlinear shifted power fit. The r^2 of the nonlinear regression is somewhat better than that of the log-linear fit. In addition, the curve of the nonlinear regression essentially passes through the center of the data. In other words, any deviation from the curve appears to be due to random variation, rather than a systematic deviation from the fitted model.

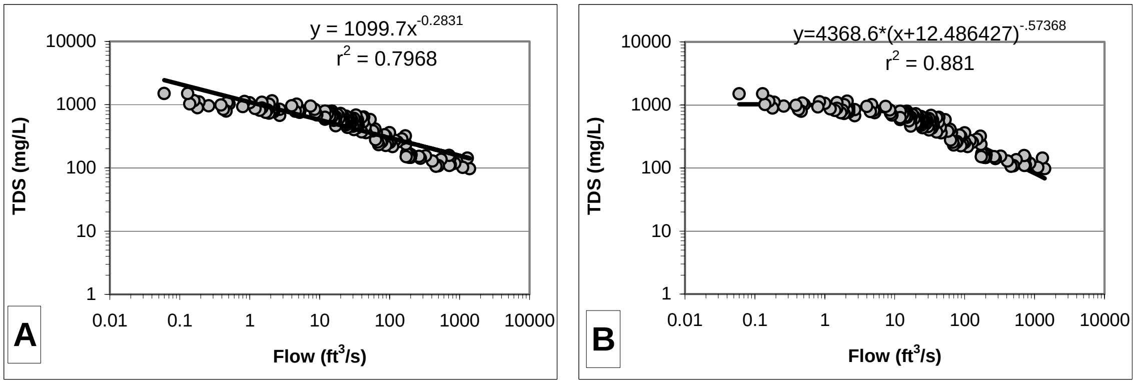


Figure 5. TDS-Flow relationships in Deer Creek below Millar Wasteway

The upper part of the curve on Figure 5B indicates that the TDS of Deer Creek becomes nearly constant at about 1,000 mg/L at flows of 5 ft^3/s or less. From the perspective of both TDS and flow, these appear to be base flow conditions. Although Deer Creek had no flow on 3 days during the water quality period of record, a review of the gage record at the site indicates that 0 flow days occur during dry years. For example, there was no flow at the Deer Creek gage for most of July and August, 1977.

Although the flow of the North Platte River at the Glenrock gage is affected by Deer Creek at times, the main reason for the poor regression appears to be a number of outliers among the TDS-flow data pairs (Figure 6A). A comparison of the outliers with data from Deer Creek on the same dates indicates that the Deer Creek flows were too low to have any significant influence in the river. On the other hand, deleting the outliers improves the TDS-flow regression greatly in comparison to the regression in Table 2 (compare with figure 6A). Although there appear to be some remaining outliers in the data set (Figure 6), these are not as obvious as the ones that were removed from the regression. The r^2 of the updated TDS-flow regression (0.526) for the Glenrock gage is still well below that of the other free-flowing river sites in Table 2, which may be a reflection of those remaining outliers.

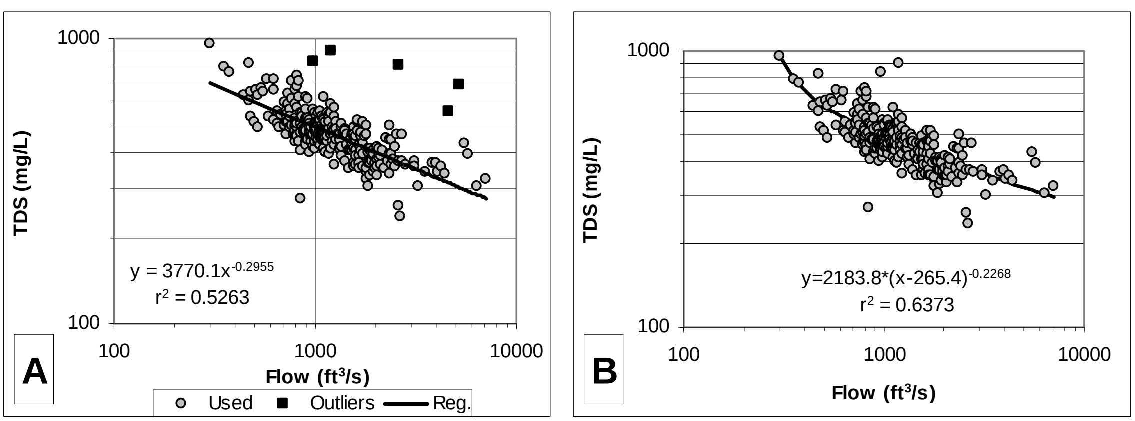


Figure 6. TDS-flow regressions based on data from the North Platte River at Glenrock
A. outliers deleted **B**. nonlinear regression

Figure 6B shows a nonlinear regression fit to the Glenrock data. The regression is based on the same model (shifted power) as was used on the Deer Creek data. The r^2 of the nonlinear regression is not greatly different from the other free-flowing river sites in Table 2. This may indicate that the remaining outliers are no more serious than the routine random spread in other data sets. The improvement in the r^2 effected by the nonlinear fit seems to be a reflection of the Deer Creek influence. Based on the above, the poor relationship between TDS and flow at the Glenrock gage is due to a combination of data outliers and the influence of Deer Creek inflows during what appears to be storm runoff events.

The r^2-values for the TDS-EC regressions in Table 2 are around 0.9, with the exception of the North Platte above Pathfinder and at Alcova, although the r^2 at the site above Pathfinder Reservoir is still 0.8. Hem (1985) indicates that the usual slope of a regression between TDS and EC is between 0.54 and 0.96 in natural waters. The regressions for the North Platte above Pathfinder and at Alcova are also the only ones with slopes outside that range. The effect of hot springs within Alcova Reservoir may be responsible for the unusual TDS-EC relationship at Alcova Dam. However, the relationship at Alcova Dam is more likely due to complex interactions among the inflows into Seminoe and Pathfinder reservoirs.

As is indicated on Figure 4, the EC (and TDS) of the Medicine Bow River is about 3 times that of the North Platte upstream from Seminoe Reservoir. The slope of the Medicine Bow TDS-EC regression has a larger slope than that of the one for the North Platte above Seminoe Reservoir. The resulting mix of water above Pathfinder Reservoir (and below Seminoe Dam) has a much smaller TDS:EC regression slope than either of the inflows. The Medicine Bow supplies a disproportionately high percentage of the salt load to Seminoe Reservoir. Although the Medicine Bow inflow accounts for 11 percent of the inflow to Seminoe, it contributes over 30 percent of the salt load. In addition, there are times when the Medicine Bow inflow forms a density flow (or interflow) within the metalimnion of Seminoe Reservoir because of its higher TDS concentration. When there is an interflow, the Medicine Bow water flows relatively unmixed through Seminoe Reservoir and would control the TDS:EC relationship at the outlet. The result that at time the TDS of the outflow is a mix of the two waters and at other times the TDS of the outlet would be more like the Medicine Bow. The net effect is a TDS:EC relationships with a slope of 0.49 and a somewhat reduced r^2 in the river above Pathfinder Reservoir (Table 2).

The North Platte River water is then mixed with the inflow from the Sweetwater River in Pathfinder Reservoir, once again mixing water with rather different TDS:EC relationships. The difference in the TDS:EC relationships are a reflection of the different ionic composition of the TDS. Divalent ions contribute more to the EC per unit of mass than univalent ions. Although the Sweetwater River only contributes about 15 percent of the inflow to Pathfinder Reservoir and its TDS is not that different from that of the North Platte inflow (Figure 3), the slope of the TDS:EC regression is greater than that of the North Platte upstream from Pathfinder Reservoir. The Sweetwater River enters the reservoir very near the dam and may flow to the outlet relatively unmixed at times. Most of the outflow from Pathfinder Dam flows directly to Alcova Reservoir through Fremont Tunnel. The net effect of all of these mixtures is the TDS:EC regression shown at Alcova Dam in Table 2.

Relationships between selenium and EC

Excessive selenium is present at toxic concentrations in the North Platte Basin. There are 2 mainstem sites for which data on water quality constituents other than EC are available, but there are no selenium data. These 2 sites include those at Casper and the Whalen Diversion Dam. In addition, the site at Glenrock has only 1 sample, and the site at the State Line has only 3 samples. It may be possible to compensate for these selenium data limitations by using a surrogate such as EC, if a relationship can be established. There are more data on EC in the North Platte Basin than for any other constituent except temperature.

Table 3 shows correlations between dissolved selenium and EC at all of the mainstem sites that were included in Table 1 and have available selenium data. There is a significant correlation between selenium and EC in the North Platte River upstream from Seminoe Reservoir, despite the fact that the vast majority of the selenium samples are below the detection limit of 1 µg/L and thus show no variation. In the river upstream from Pathfinder Reservoir, the correlation coefficient is not only not statistically significant, it is negative, indicating that as EC increases, selenium decreases. The inflow to Pathfinder is not far from Seminoe Dam. Hypothetically, there is some effect of the mixing of water from the North Platte and the Medicine Bow rivers within the reservoir that results in the odd relationship between EC and selenium downstream from Seminoe Dam. There is similarly no correlation between EC and selenium downstream from Alcova Dam, but the correlation coefficient is now positive, although just barely so (Table 3). All of this may reflect the same factors governing the TDS:EC relationships discussed above.

Table 3. Correlations between EC and selenium

Site	r	Prob. > r	n
Seminoe	0.388	0.023278	34
Pathfinder	-0.697	0.123516	6
Alcova	0.003	0.983061	42
above Poison Spider Creek	0.673	0.000063	29
Mills	0.875	< 0.000001	20
below Casper	0.856	< 0.000001	63
Glenrock	—	—	1
Orin	0.849	0.000062	15
WY-NE	0.773	0.437411	3

Beginning with the site above Poison Spider Creek, the influence of the tributaries draining the Kendrick Project becomes evident (Table 3). Bates Creek and Poison Spring Creek enter the river above the site. There is a very highly significant correlation between selenium and EC at Mills. The correlation coefficient is nearly 0.9 (Table 3). The correlation coefficient remains near 0.9 from Mills to the Orin gage just upstream from Glendo Reservoir. The strong correlation between selenium and EC, which is also a surrogate measurement of total dissolved solids (TDS), would indicate that there is a common factor affecting both of these water quality constituents.

The correlation coefficient between selenium and EC at the State Line gage is only slightly less than that for the 3 sites in the Alcova Dam to Glendo Reservoir reach of the North Platte River (Table 3). However, the correlation is not statistically significant because there are only 3 samples. Because of a lack of data, there is considerable uncertainty concerning selenium in the reach of the North Platte River between Glendo Dam and the State Line.

The NIWQP sampled many of the tributaries shown on Figure 3 for both dissolved and total selenium, with the addition of 6-mile Draw. Those results showed no significant difference in the concentration of dissolved and total selenium in the samples, as well as a very good correlation between the 2 (See *et al.*, 1992). However, those comparisons were made on lumped data sets. A similar analysis of the data to that of See *et al.* (1992) is shown in Table 4, but the data are broken down by individual watercourse. There are no significant differences in the concentrations of dissolved and total selenium at any of the sites based on paired t-tests. There are also very good correlations between the total and dissolved selenium concentrations at each of the sites.

Table 4. Relationship between dissolved and total selenium (µg/L) in 6 tributaries to the North Platte River near Casper

Stream	Diss. Se	Total Se	t	Prob. > t	r	Prob. > r
Bates Creek	5.0	5.2	-0.690	0.505852	0.907	0.000117
Poison Spring Creek	78	79	-0.637	0.534646	0.977	< 0.000001
Poison Spider Creek	57	57	-0.216	0.831711	0.952	< 0.000001
Oregon Trail Drain	282	291	-0.820	0.428272	0.980	< 0.000001
6-Mile Draw	204	205	-0.143	0.893295	0.993	0.000677
Casper Creek	77	80	-1.396	0.182960	0.991	< 0.000001

The fact that the vast majority of the selenium is in the dissolved form is consistent with the correlation between selenium and EC. Selenium is behaving as the salts of major ions, which are also the electrolytes that govern the EC of the water. If the selenium is essentially all dissolved, the vast majority of it should be in the form of selenate (SeO_4), which is chemically similar to sulfate (SO_4), one of the major ions. (Selenium and sulphur are in the same group on the Periodic Table of the elements.) For all of the above reasons, EC (or TDS) should also be a good surrogate for selenium.

Selenium is on the Wyoming list of impaired waters, *i.e.* waters that do not meet water quality standards. The selenium problem in the basin is discussed in a later section on impaired waters in the Platte River basin, including the North and South Platte basins. The purpose of the preceding was simply to provide some background for the later more regulatory aspect of water quality analysis.

Environmental Consequences

Method of analysis

The TDS analysis will be performed on the alternatives as defined in the July 2005 hydrologic simulations using the North Platte River hydrologic model. The TDS analysis will focus on changes in the TDS of North Platte reservoirs due to the demands for delivery to the Central Platte River in Nebraska, and the significance of any changes will be documented.

Most of the gages on the North Platte River have a water quality record that begins in the 1960's (Table 1). The North Platte model uses a period from 1947 through 1994. For purposes of the water quality effects analysis, a period from 1961 through 1994 was used.

The water quality data sets were constructed by extracting the TDS data from the data downloaded from NWIS as described in Table 1 above. The goal was to create a monthly TDS data set for the analysis. None of the gages had a complete monthly TDS or EC data set for the entire period of 1961 through 1994. To create the monthly TDS data set, a mean for the month was used for any month for which there was more than one TDS (or EC) observation. Where there were specific conductance data, but no TDS, the TDS:EC relationships shown in the Table 2 were used to generate a TDS concentration. For months for which there were neither TDS nor specific conductance data, the TDS concentration was calculated from the appropriate TDS-flow regression shown in Table 2, using the average flow in ft^3/s for the month. The exceptions to this include the inflows to Pathfinder and Alcova reservoirs. For the Pathfinder inflow, the Seminoe outflow was combined with the Sweetwater inflow to create a flow weighted TDS. In the case of the Alcova inflow, the Pathfinder flow-weighted TDS was used.

The monthly data sets were used to calculate a running flow-weighted average TDS based on the average residence time of water in the reservoir. The residence time was calculated from the model output by taking the average water volume in the reservoir and dividing by the average total annual inflow. This method admittedly ignores any fluctuations in the residence time due to wet and dry years, except as they influence the TDS of the inflow. The running average TDS was calculated by flow-weighting the TDS of the inflow or inflows in the case of Seminoe and Pathfinder reservoirs and summing the weighted inflow TDS and the total inflow for the period comparable to the residence time. The weighted inflow TDS was divided by the total inflow over the period. Partial months were included at the beginning of the period. To create a flow-weighted average for 1961, the calculations were initiated in 1960 to create a buffer against rapid changes in the reservoir TDS during the period used in the comparisons.

The running flow-weighted average simulates the effect of the TDS in storage in the reservoir and dampens temporal fluctuation in TDS observed in the inflows. However, the method also assumes a fully mixed condition in the reservoirs. To compensate for this, the annual TDS is also used to evaluate results. The annual TDS is represented by the geometric mean of the 12 monthly TDS values for each reservoir.

The results presented here will consist of a comparison of the annual mean TDS, plotted first as a time series and then as a cumulative frequency distribution. In addition, a worst case analysis based on the comparison of end-of-month TDS estimates for the year of peak TDS, will be presented for the Present Condition and each alternative.

Present Condition

In the TDS analysis of the North Platte reservoirs that was done for the DEIS, it was noted that there were no TDS data for the North Platte River upstream from Pathfinder Reservoir, which is also downstream from Kortes and Seminoe dams. In conjunction with the updated analysis for the FEIS, data for the site were again retrieved from the USGS National Water Information System (NWIS). There are still no TDS data in the NWIS database.

TDS can be determined in either of two ways, by evaporating a water sample to dryness and weighing the amount of salt left behind or by adding up the total of the most common individually determined dissolved solids, also called the major ions (calcium, magnesium, sodium, potassium, chloride, sulfate, and carbonate). The retrieved data included all of the major ions, except the carbonates, in the period from 1987 through 1989. However, among the analytes, was a measure of the acid neutralizing capacity of the samples collected between 1987 and 1989, a total of 12 samples. Acid neutralizing capacity is essentially the same titration procedure as is used in the determination of carbonates. The main difference is that the result is expressed as calcium carbonate, rather than as bicarbonate and carbonate. The TDS of the samples were calculated by summing the major ions listed above and adding 60 percent of the concentration of the acid neutralizing capacity (60 percent is the conversion of the molecular weight of calcium carbonate [50] to the equivalent weight of carbonate [30]). The resulting TDS data were used to calculate the regression for the North Platte River above Pathfinder that was previously presented in Table 2.

In the water quality appendix to the DEIS for the Program, the calculated TDS of Seminoe Reservoir under the simulated Present Condition was compared to measured data based on profiles collected near the dam. Those results indicated a good correlation between the measured and calculated TDS. However, the comparison also showed that the calculated TDS was significantly lower than the measured TDS. Among the reasons given for underlying causes of the differences were variation in time and space in the reservoir TDS that would not be represented in the instantaneous reservoir measurements. Averaging the TDS of the profile would account for the effect of depth, but it would not account for longitudinal variations. The calculated TDS on the other hand represents a reservoir-wide average end-of-month TDS concentration.

In the recent NWIS retrieval, the same TDS profiles from Seminoe Reservoir near the dam were included. However, profiles for the same dates from a site in the North Platte Arm of the reservoir were also retrieved. These TDS data from the North Platte Arm can provide a basis for evaluating longitudinal variation within the reservoir. Such a comparison is made in Table 5A. As can be seen in the first comparison, the TDS concentration of the North Platte Arm is significantly lower than that of the dam, although there is a good correlation between the 2 sets of data. This difference exemplifies one of the factors that could have been a factor in the difference between the measured and Present Condition calculated TDS of Seminoe Reservoir.

Table 5. Comparison of various measures of TDS (monthly average – mg/L) in Seminoe Reservoir – paired t-tests (MM = Miracle Mile)						
A. Measured TDS of various Seminoe Reservoir areas						
Location 1	Mean TDS	Location 2	Mean TDS	r	t	Prob. > t
NP Arm	275	*at Dam*	307	0.918	-3.827	0.0064842
NP Arm	275	*MM - EC*	293	-0.067	-0.730	0.4888411
at Dam	307	*MM - EC*	293	-0.099	0.471	0.6485351
B. Measured and calculated TDS in Seminoe Reservoir						
Source 1	Mean TDS	Source 2	Mean TDS	r	t	Prob. > t
Reservoir	291	*Calculated*	280	0.922	1.252	0.2506302
MM - EC	292	*Calculated*	269	-0.169	1.401	0.1947497

Table 5A also presents a comparison between the measured TDS in the reservoir and the TDS of the North Platte River in the Miracle Mile, *i.e.* the North Platte River above Pathfinder Reservoir. The TDS of the Miracle Mile was calculated from the TDS-EC regression shown in Table 2. There are EC data from 1969 to 1989 in the Miracle Mile reach of the river. The TDS concentration of the Miracle Mile shows no statistically significant difference from neither that of the North Plate Arm nor that of the site near the dam. The average TDS of the Miracle Mile is intermediate between the 2 sites in Seminoe Reservoir. In addition, there is no correlation between the TDS of the Miracle Mile and the concentrations in the reservoir. The lack of a correlation is probably a reflection of the depth distribution in the reservoir and the layers from which the release to the river is withdrawn.

Table 5B shows a statistical comparison between the average TDS of the reservoir and the calculated TDS. The data are also plotted on Figure 7. The reservoir average represents the measurements taken in the profiles from the North Platte Arm and near the dam. In the case of August 1978, when 2 sets of measurements were made, the measured data also includes the average of the 2 dates. Unlike the comparison in the DEIS appendix, there is no significant difference in the measured and calculated TDS (Table 5B). The correlation between the measured and calculated TDS is slightly better than the one shown in the DEIS.

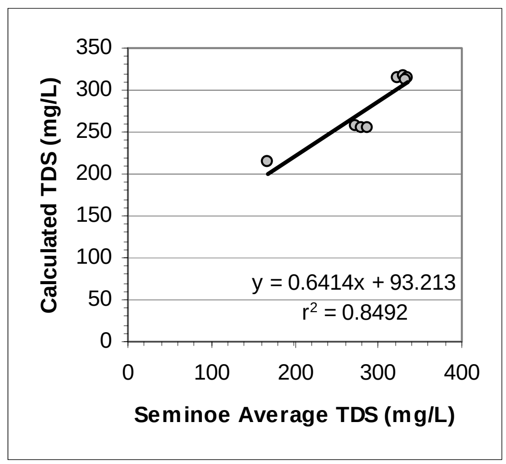


Figure 7. Average and Present Condition calculated TDS in Seminoe Reservoir

Table 5B also shows a comparison between the calculated TDS concentration of the reservoir and that of the Miracle Mile. The results are similar to the comparison between the Miracle Mile TDS and the measured TDS of the reservoir. There is no significant difference in TDS, but there is no correlation either. Based on these results, the calculated TDS may be representative of the TDS of the North Platte River above Pathfinder, particularly over a long-period of time.

As was noted in the Methods section above, the alternatives comparison will be based on average annual TDS. Figure 8 shows a time series plot of the average annual TDS of the Miracle Mile and the calculated TDS of Seminoe Reservoir. As can be seen from the plot, the calculated TDS is lower than the measured TDS during many of the years. During the period used for the above comparison (1976-78), there is good agreement, but there is a large disparity in the years immediately preceding that period. Of the causes of the disparity between the measured and calculated TDS that were given in the DEIS, the one that remains is the invariant residence time in the reservoir. This factor seems to remain, but the results may not be a bad as they appear on Figure 8.

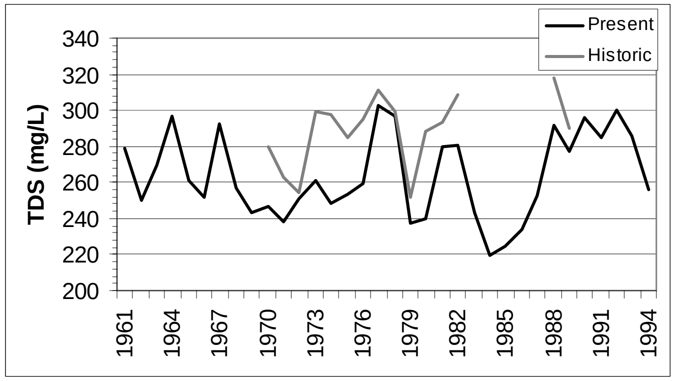


Figure 8. Comparison of historic and Present Condition annual mean TDS in Seminoe Reservoir

Figure 9 shows another comparison between the calculated TDS and the measured TDS of the Miracle Mile. The relationship is rather poor, when all of the data are included (see the regression in the lower left hand corner on Figure 9), but when the outlier is dropped, there is a reasonable correlation between the measured and calculated TDS most of the time (equivalent r = 0.81). However, as is shown by the slope of the regression equation, the calculated TDS is about 89 percent of the measured TDS on the average.

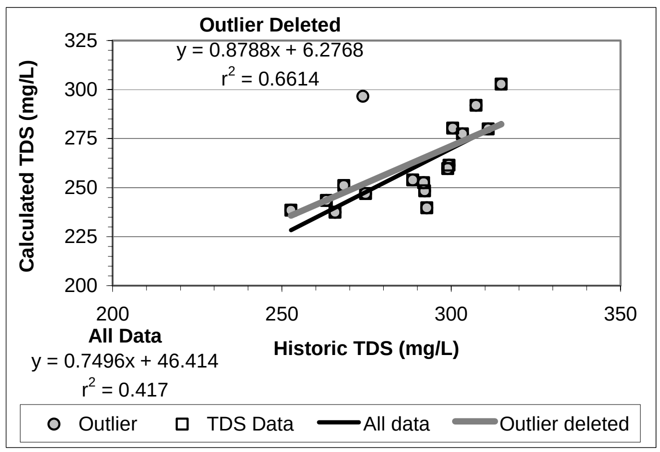


Figure 9. Comparison of historic Miracle Mile and Present Condition annual average TDS in Seminoe Reservoir

The comparisons based on Table 5 indicates that the monthly TDS gives a reasonable comparison to the historic data. In addition to the comparison of the annual TDS, the alternatives comparison also includes a monthly comparison during the year of peak TDS. This is represented by 1977 in the record. This peak can be seen on Figure 8 in the annual data plot for the Present Condition. A comparison of the 1977 Present Condition calculated and the historic TDS from the TDS-EC regression is shown on Figure 10. The calculated and historic TDS do not show much agreement. The peak historic TDS occurred in February, while the peak calculated TDS occurred in April. However, if the calculated TDS is shifted back 2 months, there is very good agreement between the

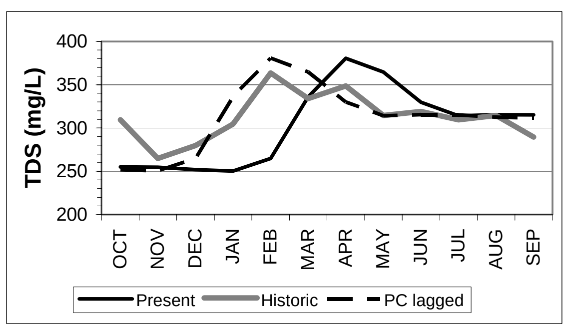


Figure 10. Comparison of Present Condition and historic outflow dry year (1977) monthly TDS from Seminoe Reservoir

observed and calculated TDS (Figure 10). The lagging effect shows how the invariant residence time in the reservoir affects the monthly TDS. In the dry year comparison, the peak occurs 2 months late. Under other circumstances, the result could go the other way with the peak TDS occurring earlier. The peak TDS in the reservoir should occur prior to runoff, before the large volume of relatively dilute water enters the reservoir. Because the invariant residence time is used in the overall analysis, the calculated TDS can still be considered only a qualitative measure of the Present Condition and the effects of the alternatives on the TDS of the North Platte Basin reservoirs as evaluated in the FEIS should be considered qualitative as well.

In addition to the reservoirs, there is also a river site that is used to evaluate the effects of the Program on the TDS of the North Platte Basin in Wyoming. The evaluation is based on the regression shown in Table 2 for the North Platte River at the Orin gage. A comparison between the calculated TDS at the Orin gage under the Present Condition and the historic TDS at the site is shown on Figure 11.

There is considerably more variation in the historic data than in the Present Condition TDS on Figure 11. Once again, the 2 sets of data represent different conditions. The historic data are instantaneous measurements, while the Present Condition TDS represents a monthly average. Fewer spikes would be expected in an average than in a spot measurement as is represented by the historic data. Unlike the calculated reservoir TDS, the calculated Present Condition TDS at the Orin gage exceeds the historic TDS on most occasions. Because of the difference between the historic and Present Condition, the impact assessment for the river site should also be considered qualitative.

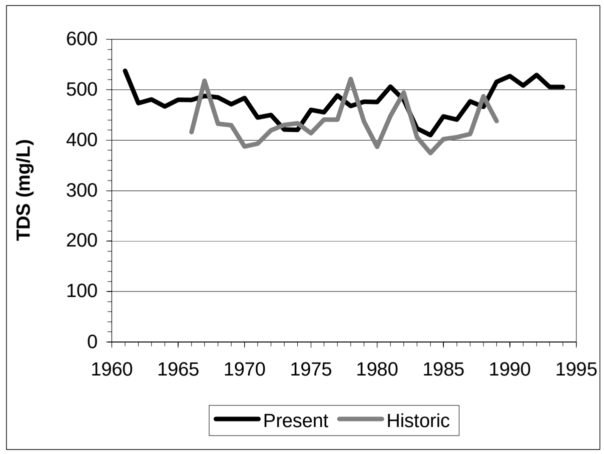


Figure 11. Present Condition and historic TDS of the North Platte River at Orin, Wyoming

Effects of Alternatives

The presentation of the effects of the Program alternatives on the North Platte River in Wyoming will work its way from Seminoe Reservoir to the State line. The presentation will be made site-by-site for each of the 4 alternatives, rather than grouping the sites and evaluating each alternative. Showing the results site-by-site allows for a clearer presentation of the alternatives comparisons.

Seminoe Reservoir

Table 6 shows a comparison of the mean annual TDS of each of the alternatives with that of the Present Condition for the period 1961 through 1994. In Table 6, the alternatives comparison columns reflect the number of years in which there are increases in TDS relative to the Present Condition by an alternative (Alt. > PC), and the number of years in which there are decreases in TDS (PC > Alt.). The results shown in Table 6 are from a series of Wilcoxon-signed ranks tests. It should be noted that in any year in which there is no difference in TDS, that year is not included in the calculation of the test statistic, Z. The Wilcoxon-signed ranks test is a nonparametric (distribution-free) equivalent to a paired t-test. For the results of a paired t-test to be statistically acceptable, the differences between the paired data must be normally distributed. Because this is not always the case, the nonparametric equivalent was used. The Wilcoxon-signed ranks test will be used to compare the TDS results for all sites in the North Platte Basin.

Table 6. Comparison of the TDS of Seminoe Reservoir under the Present Condition (PC) and with that of the other Platte River RIP Alternatives

Alternative	Alt > PC	PC > Alt.	Z Stat.[1]	Prob. > Z
Gov. Comm.	28	6	3.82	0.000133
Water Emphasis	29	5	4.56	0.000005
Full Water Leasing	2	32	-5.03	< 0.000001
Wet Meadow	32	2	5.00	0.000001

[1] Z = (Sum of signed ranks)/square root(sum of squared ranks)

Based on the results in Table 6, there is a statistically significant difference between the mean annual TDS of each of the alternatives and that of the Present Condition. For 3 of the 4 alternatives, there is an increase in TDS, as represented by a positive test statistic, Z. The formula for Z is given in the footnote to Table 6. The 3 alternatives that show an increase in TDS include the Governance Committee, Water Emphasis, and Wet Meadow Emphasis alternatives. The decrease in TDS would occur under the Full Water Leasing Alternative.

The Wilcoxon-signed ranks test evaluates the number of years in which TDS increases against the number years in which it decreases. The expected split is 50:50. A statistically significant Z indicates that the result differs from the expected 50 percent for each. However, this type of result does not show the magnitude of the differences. A large number of small increases or decreases can give a statistically significant result. It should be noted that a paired t-test, which does evaluate the magnitude of the differences, can give the same type of result. If all of the differences are small, the variance of the differences, which is the denominator in the calculation of the test statistic (t), will also be small, possibly giving a large (or statistically significant) value of t.

A comparison between the TDS of each of the alternatives with that of the Present Condition is shown on Figure 12. The plots on Figure 12 consist of a set of time series plot of the annual average TDS of Seminoe Reservoir with each of the alternatives on the left-hand side. Each of these plots includes the Present Condition to facilitate a comparison. The right-hand side of Figure 12 includes a set of cumulative frequency plots of the monthly TDS of each alternative plotted along with that of the Present Condition. Both sets of plots include the calculated TDS for the period 1961 through 1994. The plots indicate that the difference between the TDS of the Present Condition and the alternatives is small, with the maximum difference of 10 mg/L. The published USGS data in the range of TDS from which the data sets used in the alternatives comparison were developed are rounded to the nearest 10 mg/L. The largest increase in the average annual TDS in Seminoe Reservoir would occur with the implementation of the Wet Meadow Emphasis Alternative. This increase amounts to 10 mg/L in 1963 (Figure 12). The median increases in TDS among the 3 alternatives with a significant increase range from 2 mg/L (Governance Committee Alternative) to 5 mg/L (Wet Meadow Emphasis Alternative). Although the results indicate a statistically significant increase (or decrease) based on the changes in average annual TDS, these changes would be so small as to be virtually unnoticeable.

The largest increase in the mean annual TDS by the Governance Committee Alternative relative to the Present Condition was 7 mg/L, which occurred in several years (1961, 1981, 1987), although 1977 was the year of the peak TDS for all of the alternatives in Seminoe Reservoir. There was an increase of 6 mg/L in the annual mean TDS in 1977. The maximum increase in the mean annual TDS of Seminoe Reservoir with the Water Emphasis Alternative was 8 mg/L, which occurred in 1961 and 1981, while the maximum increase in the mean annual TDS with the Full Water Leasing Alternative was less than 1 mg/L. As should be evident by now, the effects of the Full Water Leasing Alternative are completely different from the other 3 alternatives, and its largest increase (if it can be called an increase) occurred in 1966 and 1987. In other words, the largest increase in TDS does not coincide with the occurrence of the peak TDS in Seminoe Reservoir for any of the alternatives relative to that of the Present Condition. Nevertheless, to be consistent with the description of the Present Condition and historic data comparison above, a monthly comparison of the TDS of Seminoe Reservoir between each of the alternatives and the Present Condition during 1977 is presented on Figure 13. The data are plotted on the basis of water year 1977, *i.e.* October through September.

The monthly TDS of the Full Water Leasing Alternative is virtually overlain on that of the Present Condition (Figure 13). The only apparent deviation occurs during May, when the TDS of the Full Water Leasing Alternative is slightly lower (23 mg/L) than that of the Present Condition. Alternatively, the plots of the monthly TDS of the other 3 alternatives appear to be copies of each other. In each case, the TDS of the alternative is overlain on that of the Present Condition in the fall and early winter and again in the late summer. However, each alternative is about 70 mg/L higher than the Present Condition TDS February and about 50 mg/L higher during March. These differences appear somewhat similar to what was shown earlier on Figure 10 in the comparison between the historic and Present Condition TDS. In that case, the annual pattern of the 2 sets of monthly TDS values were offset by about 2 months. The difference between the annual distribution of monthly TDS of the Present Condition and the Governance Committee, Water Emphasis, and Wet Meadow Emphasis alternatives is explored more fully on Figure 14, using the Governance Committee Alternative as an example.

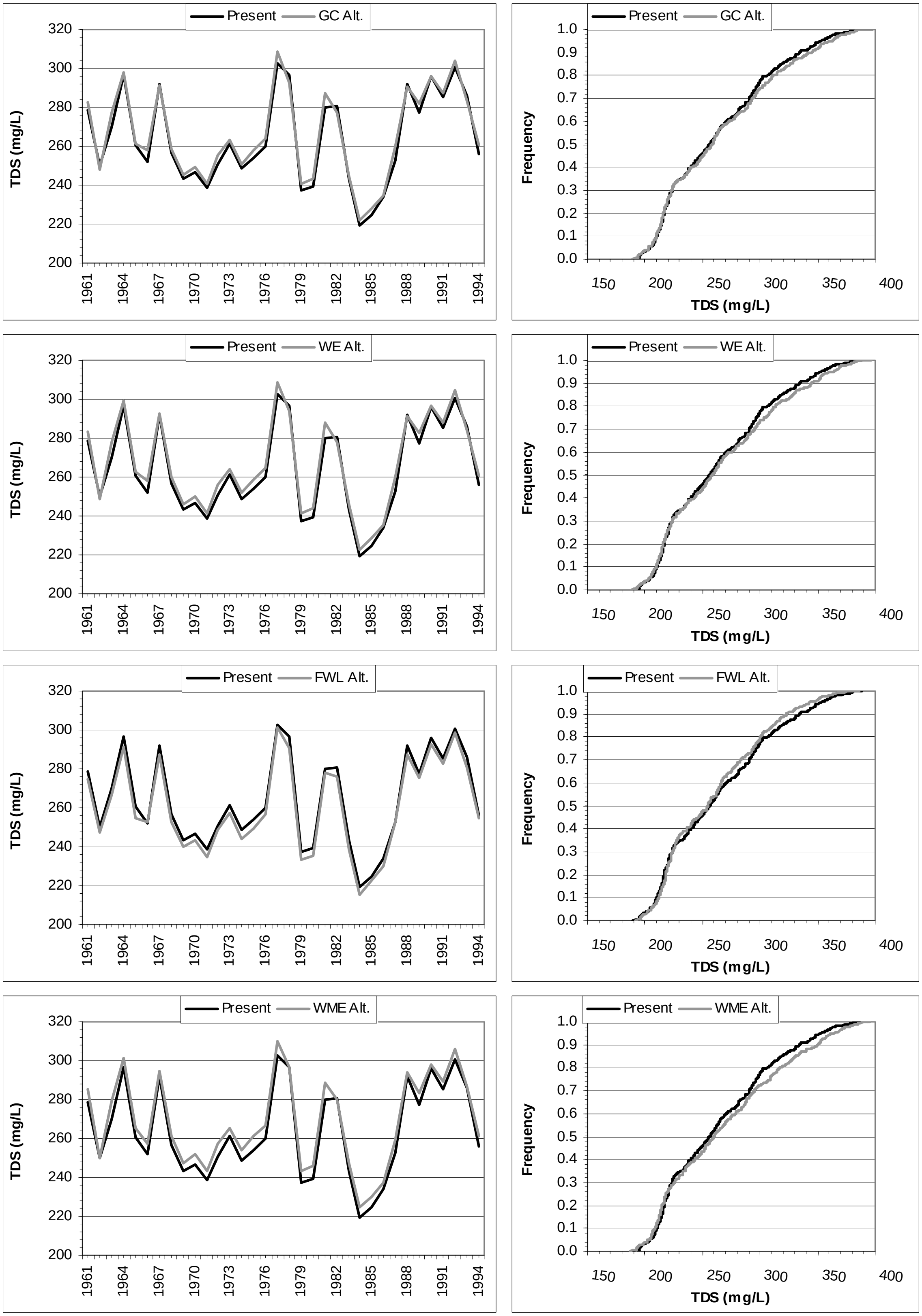


Figure 12. Comparison of the mean annual TDS and the cumulative frequency of the monthly TDS of the Present Condition and each alternative in Seminoe Reservoir

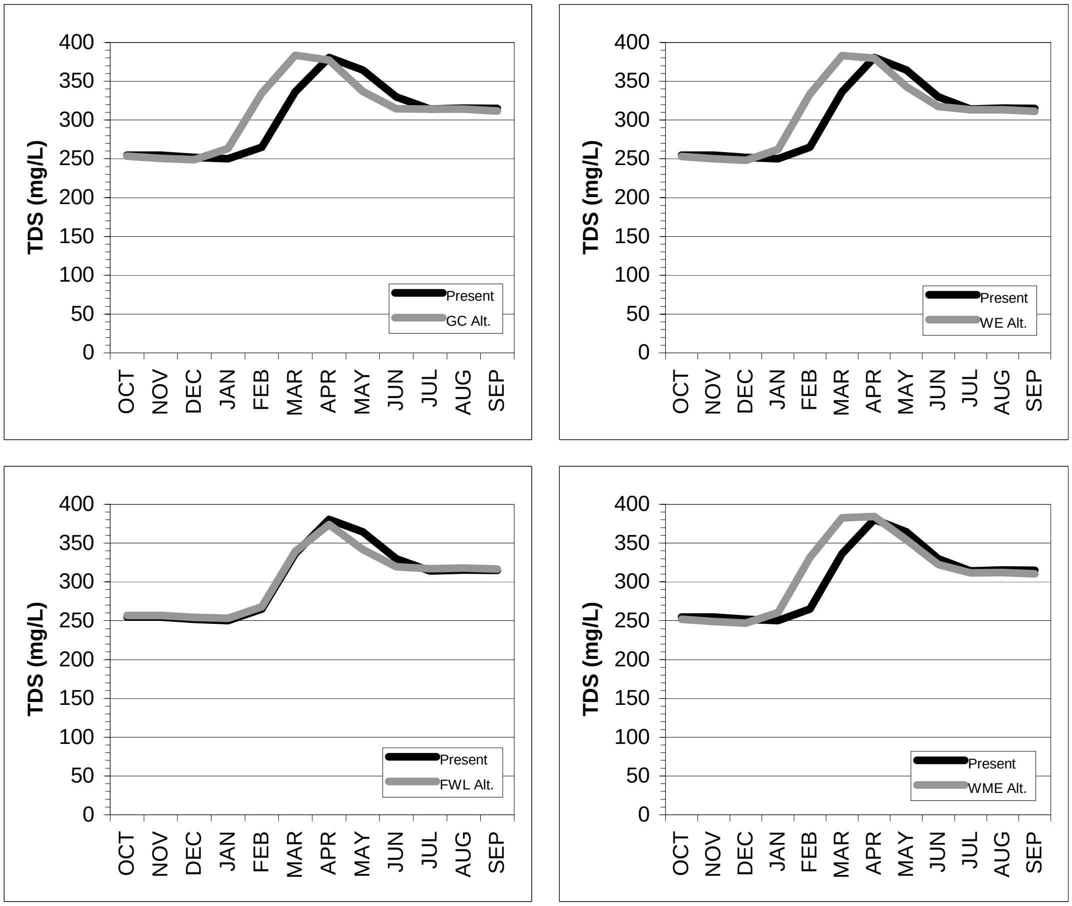


Figure 13. Monthly TDS of Seminoe Reservoir with each of the Program alternatives and the Present Condition

Figure 14 was developed by shifting the Governance Committee Alternative monthly TDS by 1 month relative to that of the Present Condition. The result of the shift is that the 2 sets of TDS data are virtually overlain. This result essentially confirms that much of the difference in the TDS of the alternatives relates to the invariant residence time used to calculate the monthly TDS. This result also reinforces the use of the annual mean TDS for the comparison of the alternatives. Although the annual mean will be influenced by the peak TDS, the mean should be affected by about the same degree for all alternatives. Furthermore, the use of a geometric annual mean, rather than an arithmetic mean, reduces the influence of the peak monthly TDS on the annual mean TDS greatly. For the remaining sites, the alternatives comparison with the Present Condition will be made solely on the basis of the mean annual TDS.

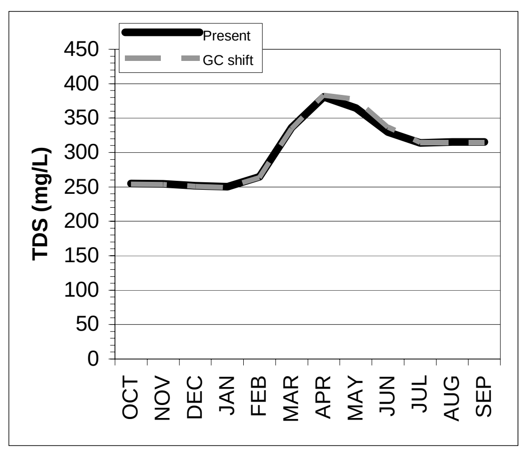


Figure 14. Monthly TDS of the Governance Committee Alternative shifted 1 month relative to the TDS of the Present Condition

Pathfinder Reservoir

Comparisons of the TDS of the various alternatives with that of the Present Condition are shown in Table 7. The majority of the water supply to Pathfinder Reservoir comes from Seminoe Reservoir. As a consequence, the wide range in the inflow typical in Seminoe Reservoir does not occur in Pathfinder. A comparison of the effects of the alternatives in Pathfinder Reservoir would reflect the drawdown and a decreased capability of the reservoir to buffer changes. Because of the effects of Seminoe Reservoir, the reduced buffering capability is less of a factor in controlling TDS. This is reflected in the results shown in Table 7. In Pathfinder Reservoir, there is no statistically significant difference between the TDS concentration of the Present Condition and that of either the Governance Committee or the Water Emphasis alternative. Of note is that the number of years in which the mean annual TDS of Pathfinder Reservoir decreases exceeds the number of years in which it increases (Table 7). There would still be a significant increase in the number of years of increased TDS with the Wet Meadow Alternative. There would be no years in which the TDS under the Full Water Leasing Alternative exceeded the TDS of the Present Condition.

Table 7. Comparison of TDS under the Present Condition (PC) and with the other Platte River RIP Alternatives

Alternative	Alt. > PC	PC > Alt.	Z Stat.	Prob. > Z
Gov. Comm.	13	21	-1.55	0.121807
Water Emphasis	17	17	0.01	0.993180
Full Water Leasing	0	34	-5.09	< 0.000001
Wet Meadow	24	10	2.62	0.008682

Figure 15 shows plots of the mean annual TDS of the Present Condition and each of the alternatives. The set of plots on the left are the mean annual TDS concentrations. As was the case with the Seminoe data on Figure 12, the right-hand set of plots is again the cumulative frequency distributions of the monthly TDS data. Although it was stated above that monthly TDS data would not be used because the way in which the TDS was calculated was more of a factor than any real effect of the alternatives. However, the use of cumulative frequency distributions eliminates the lagging effect of residence time. For example, the maximum monthly difference in TDS between the alternatives is 37 mg/L for the Water Emphasis and Wet Meadow Emphasis alternatives and 27 mg/L for the Governance Committee Alternative. The maximum monthly differences in TDS from the Present Condition range between 8 and 11 mg/L for the 3 alternatives in the cumulative frequency distributions. As will be shown momentarily, these differences are more in line with those of the mean annual TDS data.

The maximum mean annual TDS in Pathfinder Reservoir occurs in 1978 with Present Condition and all of the alternatives. This is a 1 year lag in the occurrence of the peak TDS from what had been shown in Seminoe Reservoir. Although it may be something of a coincidence, the total storage and the average annual inflow for the period, 1961 through 1994, are approximately the same (1, 017 and 996 thousand acre-feet respectively), giving a water residence time of approximately 1 year. The effects of storage and mixing in Seminoe Reservoir on the TDS of the Pathfinder Reservoir inflow could account for the delay in its peak TDS.

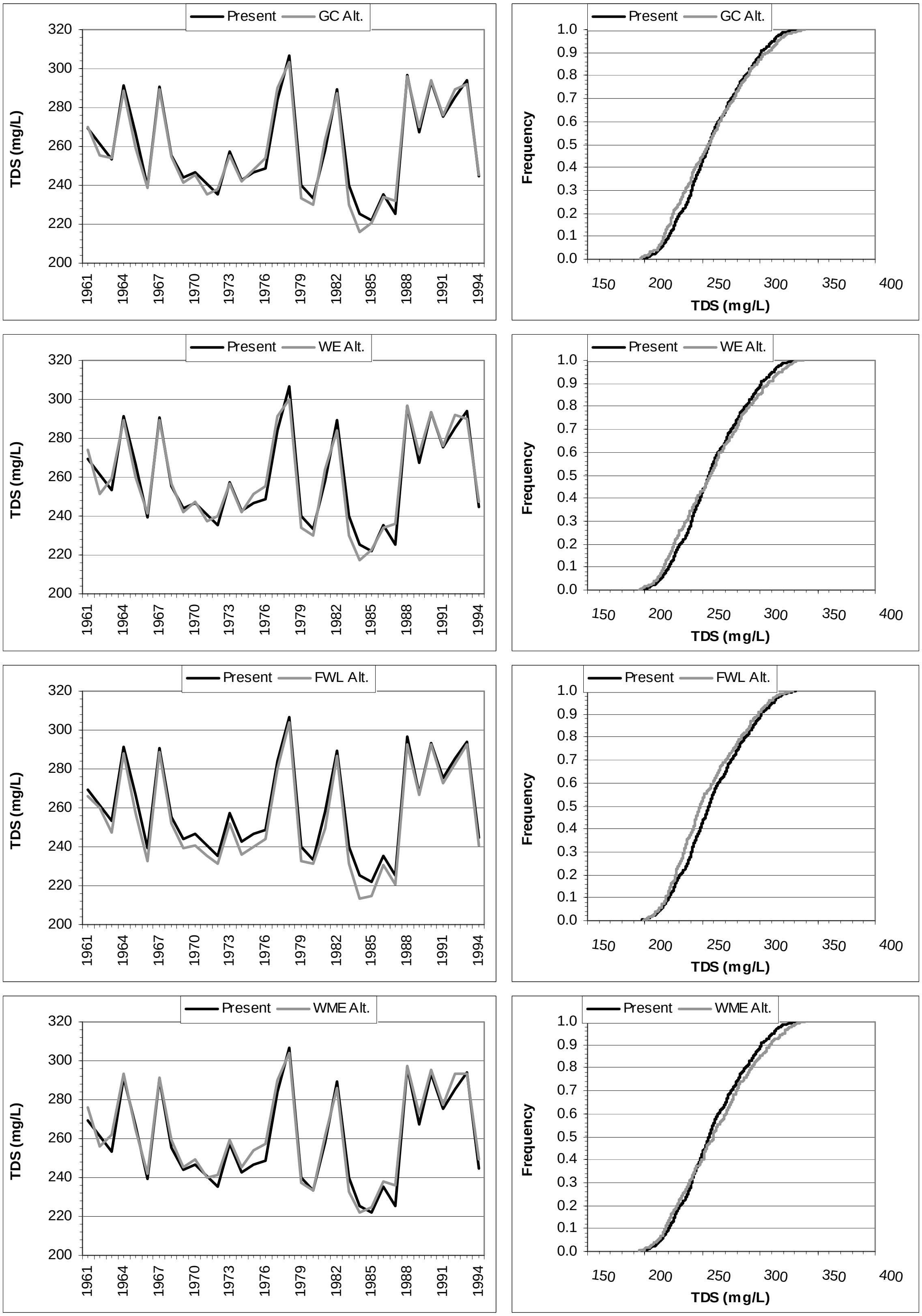


Figure 15. Comparison of the mean annual TDS and the cumulative frequency of the monthly TDS of the Present Condition and each alternative in Pathfinder Reservoir

The maximum increase in the mean annual TDS in Pathfinder Reservoir are between 7 and 11 mg/L. As shown in Table 7, there is no year with a projected increase under the operations of the Full Water Leasing Alternative. The Governance Committee Alternative would cause the 7 mg/L increase while the Water Emphasis and Wet Meadow Emphasis alternatives would each cause a maximum projected increase in the mean annual TDS relative to the Present Condition. Although the maximum increases for the 3 alternatives are slightly larger than those in Seminoe Reservoir, the Pathfinder Reservoir maximum TDS increases still round to 10 mg/L, the minimum difference in the rounded TDS of the NWIS data. Even the maximum annual increases in TDS in Pathfinder Reservoir would be virtually unnoticeable with any of the alternatives.

Alcova Reservoir

Alcova Reservoir acts as an afterbay for Fremont Canyon Powerplant. Even on a daily basis, the inflows and outflows are equal. Consequently, any changes in Pathfinder Reservoir would be expected to be passed directly to Alcova Reservoir. The results in Table 8 would indicate that this is the case. There is no significant difference between the mean annual TDS of the Present Condition and that of either the Governance Committee or Water Emphasis alternatives, a significant decrease with the Full Water Leasing Alternative, and a significant increase with the Wet Meadow Emphasis Alternative, just as was the case in Pathfinder Reservoir (compare tables 7 and 8). There are some slight differences between the alternative comparisons in the 2 reservoirs. The most notable difference is that there are fewer years in which there is a decrease in the mean annual TDS from the Present Condition for the Governance Committee, Water Emphasis, and Wet Meadow Emphasis alternatives in Alcova Reservoir. The differences amount to 1 or 2 years and do not change the results with respect to the statistical significance of the alternatives comparison to the Present Condition based on the Wilcoxon tests presented in Table 8.

Table 8. Comparison of TDS under the Present Condition (PC) and with the other Platte River RIP Alternatives

	Alt. > PC	PC > Alt.	Z Stat.	Prob. > Z
Gov. Comm.	15	19	-1.12	0.262790
Water Emphasis	18	16	0.01	0.993180
Full Water Leasing	0	34	-5.09	< 0.000001
Wet Meadow	25	9	3.00	0.002696

Figure 16 shows plots of the mean annual TDS in Alcova Reservoir for the Present Condition and each of the alternatives – the time series on the left. As was the case with Pathfinder Reservoir, the peak mean annual TDS occurs in 1978 under the Present Condition and with all of the alternatives. Although Table 8 indicates that there are many years in which the mean annual TDS is increased with 3 of the 4 alternatives, at the magnitude of the increases, they are difficult to see on the plots. Years in which the TDS decreases are much easier to discern on the plots. The largest decreases in TDS are greater than the maximum increases in TDS, except for the those of the Wet Meadow Emphasis Alternative. The maximum increases in TDS are 6 and 8 mg/L for the Governance Committee and Water Emphasis alternatives respectively, while the greatest decreases are 11 mg/L for each. Once again there are no increases for the Full Water Leasing Alternative, but a maximum decrease of 13 mg/L. The Wet Meadow Emphasis

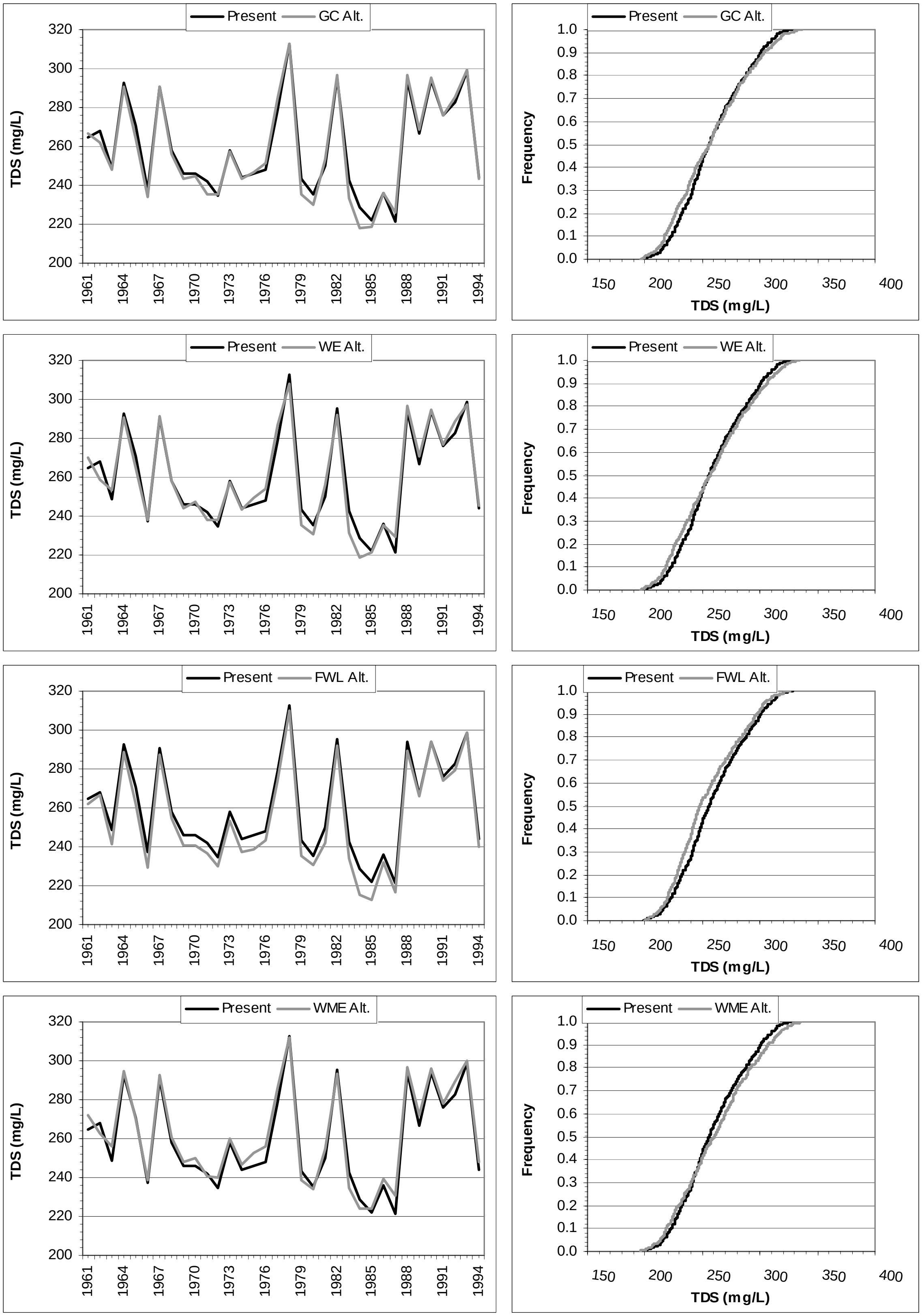


Figure 16. Comparison of the mean annual TDS and the cumulative frequency of the monthly TDS of the Present Condition and each alternative in Alcova Reservoir

Alternative has a maximum increase of 9 mg/L, while the equivalent decrease is 8 mg/L. Once again, these maxima all round to 10 mg/L and would be unnoticeable, regardless of whether or not there is a statistically significant change in TDS in Table 8.

Summary of TDS changes in the Upper North Platte Reservoirs

The preceding indicates that changes in TDS in Seminoe, Pathfinder, and Alcova reservoirs are relatively small. Those analyses were based on either annual averages or the average monthly TDS results. As was noted in the comparison of the monthly average for the Governance Committee Alternative and the Present Condition, the differences in the spring can be somewhat large. The differences in individual months can be even larger. The maximum differences for the three reservoirs in any one month, along with the net change based on monthly data, are shown in Table 9.

Table 9. Changes in TDS in comparison with that of the Present Condition (all in mg/L)

Reservoir	Alternative	Maximum Decrease	Net Change	Maximum Increase
Seminoe	Governance Committee	-58	2	100
	Water Emphasis	-49	3	100
	Full Water Leasing	-50	-3	7
	Wet Meadow Emphasis	-27	5	99
Pathfinder	Governance Committee	-25	-1	27
	Water Emphasis	-31	0	37
	Full Water Leasing	-24	-5	11
	Wet Meadow Emphasis	-29	2	37
Alcova	Governance Committee	-22	-1	21
	Water Emphasis	-26	0	31
	Full Water Leasing	-23	-5	7
	Wet Meadow Emphasis	-24	2	33

At Seminoe Reservoir, both the greatest increase and the greatest decrease in TDS in any one month relative to the Present Condition are shown by the Governance Committee Alternative, although the increase is matched by the Water Emphasis Alternative (Table 9). To put things into the perspective of the preceding analysis, the net change in the TDS in Seminoe Reservoir is the smallest increase of the three alternatives that show an increase. The Full Water Leasing Alternative is projected to show a net decrease in TDS in Seminoe Reservoir. That decrease in TDS is associated with the smallest overall range in TDS, primarily because the maximum increase is the smallest of any of the alternatives. In the same vein, the Wet Meadow Emphasis Alternative shows the largest projected net increase, although it is only 5 mg/L. The Wet Meadow Emphasis Alternative shows about the same maximum increase in TDS as the Governance Committee and Water Emphasis alternatives, but that is associated with the smallest maximum decrease in TDS of any of the alternatives (Table 9). The point is that the there may be some relatively large changes in the TDS of Seminoe Reservoir at times, but these are transient, and the net changes are quite small.

The changes in TDS in Pathfinder and Alcova Reservoirs are smaller than those in Seminoe (Table 9). The maximum decrease in TDS in Pathfinder Reservoir is shown by the Water Emphasis Alternative, which also shows the largest increase in TDS, although the maximum increase is matched by the Wet Meadow Emphasis Alternative. The Wet Meadow Emphasis Alternative is the only alternative that shows a net increase in TDS in Pathfinder Reservoir. Interestingly, the Full Water Leasing Alternative shows the largest net decrease in TDS, but the smallest maximum decrease. The Full Water Leasing Alternative once again shows the smallest maximum monthly increase in TDS in Pathfinder Reservoir, but that increase is larger than the one in Seminoe Reservoir. The Governance Committee Alternative is intermediate in all three categories of TDS change in Pathfinder Reservoir.

The maximum TDS changes from the Present Condition in Alcova Reservoir due to ail of the alternatives are slightly smaller than those in Pathfinder Reservoir (Table 9). Although the net changes in TDS in Alcova Reservoir are identical to those in Pathfinder Reservoir, the alternatives that show the maximum changes are different. The maximum monthly increase in TDS is shown by the Wet Meadow Emphasis Alternative alone. The maximum monthly decrease in TDS in Alcova Reservoir is shown by the Water Emphasis Alternative, as was the case in Pathfinder Reservoir. The Governance Committee Alternative again shows intermediate changes in all three categories of TDS change from the Present Condition, as was the case in Pathfinder Reservoir.

North Platte River at the Orin Gage

The site at the Orin gage will reflect the influence of the Program on the TDS of the North Platte River downstream from the saline inflows between Alcova Dam and Casper. Although the Orin gage is a considerable distance downstream from Casper, it is the only site with operations model output that can be used to evaluate changes in flow between Alcova Dam and Glendo Reservoir. Even if this were not the case, the fact that the relationship between TDS and flow at the Orin gage is so much better than that of the next upstream gage at Glenrock would make Orin the site of choice for evaluating the effects of the Program alternatives on TDS.

There was a trend toward a decreasing influence of the Program alternatives on the TDS of the North Platte reservoirs as one proceeded downstream from Seminoe Reservoir. At Orin, the trend is somewhat reversed. Table 10 shows a summary of the significance of the effects of the Program alternatives on TDS relative to the Present Condition. At Orin, the Z-statistic for all of the tests is negative, indicating a decrease in TDS relative to the Present Condition. The decreases due to all of the alternatives except for the Full Water Leasing Alternative are statistically significant. This is also a break from the results at the reservoirs upstream, in that the Full Water Leasing Alternative produced a significant reduction in TDS at those sites. The reason behind the change is that the Full Water Leasing Alternative was

Table 10. Comparison of Program Alternatives TDS with the Present Condition at the Orin gage on the North Platte River

Alternative	PC > Alt.	Alt. > PC	Z	Prob. > Z
Governance Committee	24	7	-3.130	0.001748
Water Emphasis	27	7	-3.704	0.000212
Full Water Leasing	19	14	-1.753	0.079637
Wet Meadow Emphasis	27	5	-3.732	0.000190

formulated to maintain water levels in reservoir to the greatest extent possible. River flows accrue from a variety of sources, not just the storage reservoirs on the mainstem of the North Platte River. Consequently, flow changes downstream from the major storage reservoirs, Seminoe and Pathfinder, are not as great as with the other 3 alternatives, which provide additional flow and dilution of saline inflows. In other words, in the process of drawing down the reservoirs to a greater extent, the Governance Committee, Water Emphasis, and Wet Meadow Emphasis alternatives provide increased river flow to dilute the salt loadings between Alcova Dam and Casper.

As was the case with the reservoirs, the comparison of alternatives to the Present Condition as presented in Table 10 does not tell how large the reductions in TDS may be, only that there are more decreases than increases. Figure 17 shows a comparison of the mean annual TDS for each of the alternatives and the Present Condition.

The differences in TDS between the Governance Committee Alternative and that of the Present Condition is most evident in the period from early 1960s through the early 1980s. Following the very wet years of 1983-84, there is little difference between the 2 sets of TDS data until the last year in the record, 1994, when the largest decrease in TDS from the Present Condition (22 mg/L) is observed. Although Table 10 indicates that there are 7 years in which the TDS of the Governance Committee Alternative would be greater than the TDS of the Present Condition, these are somewhat difficult to see on Figure 17. Those that can be seen appear small. However, there is an increase of 20 mg/L that occurs in 1971 (Figure 17).

The Full Water Leasing Alternative shows a similar pattern to the Governance Committee Alternative in the last part of its record. Although the maximum decrease for the Full Water Leasing Alternative also occurs in 1994, that decrease is somewhat smaller than the one shown by the Governance Committee Alternative, *i.e.* 14 mg/L. In addition, the TDS of the Full Water Leasing Alternative also tracks that of the Present Condition much better than the Governance Committee Alternative. As a consequence, there is no significant change in TDS between the Present Condition and the Full Water Leasing Alternative, as is shown in Table 10.

The Water Emphasis and the Wet Meadow Emphasis alternatives show similar distributions of TDS relative to the Present Condition (Figure 17). These 2 alternatives show differences from the Present Condition throughout the period of record shown on Figure 17, but the Wet Meadow Emphasis Alternative shows a more significant difference from the Present Condition than the Water Emphasis Alternative (Table 10). The difference is in the fewer years in which there are increases in TDS by the Wet Meadow Emphasis Alternative, *i.e.* 5 as opposed to 7 for the Water Emphasis Alternative. The number of years of decrease is the same for the 2 alternatives. This means that there are a total of 34 years in the comparison with the Present Condition for the TDS of the Water Emphasis Alternative, while there are only 32 year in the equivalent analysis for the Wet Meadow Emphasis Alternative. The difference in the total number of years is a result of years in which there is no difference between the TDS of an alternative and that of the Present Condition. Such occurrences are not included in the analysis in the Wilcoxon-signed ranks test.

The range in TDS on the *y*-axis on Figure 17 is from 400 to 560 mg/L. The equivalent range in TDS on Figure 16 for Alcova Reservoir is from 200 to 320 mg/L. The higher TDS on the *y*-axis

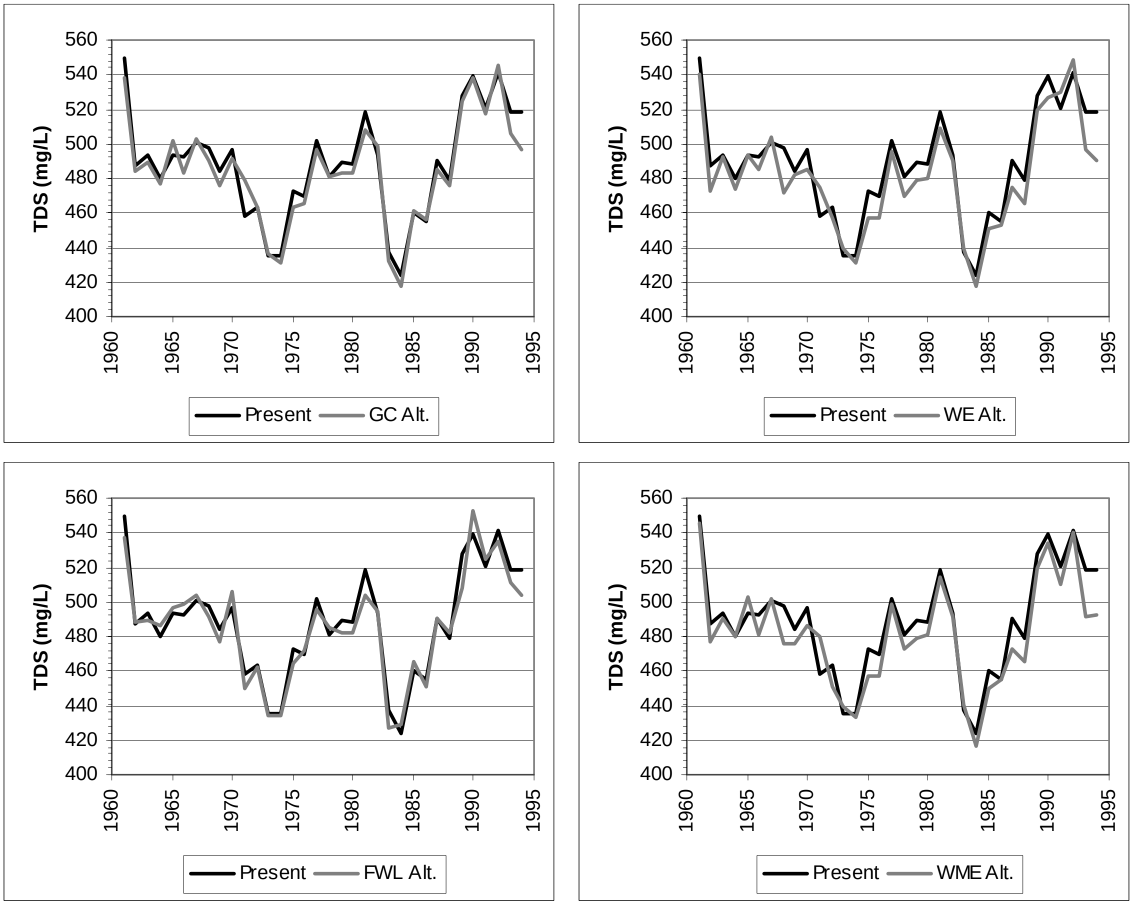


Figure 17. Mean annual TDS of the North Platte River at Orin, Wyoming, under the Present Condition and each of the 4 Program alternatives

of Figure 17 reflects the effects of the saline inflows from the tributaries between Alcova Dam and Casper. The results in Table10 indicate that the Wet Meadow Emphasis Alternative would have the greatest positive effect on the North Platte River in that reach in terms of diluting those salt loadings into the Alcova-Casper reach of the river, but the Water Emphasis and Governance Committee alternatives would also lead to a significant improvement over the Present Condition.

Glendo Reservoir

Table 11 shows a comparison between the mean annual TDS of the Program alternatives with the TDS of the Present Condition. Although all of the values of Z for the Wilcoxon-signed ranks tests are negative, indicating a decrease relative to the Present Condition, none of them are statistically significant. In other words, the alternatives would lead to a slight, but not significant decrease in TDS relative to the Present Condition.

Table 11. Comparison of TDS under the Present Condition (PC) and with the other Platte River RIP Alternatives

Alternative	Alt. > PC	PC > Alt.	Z Stat.	Prob. > Z
Gov. Comm.	13	21	-0.71	0.478011
Water Emphasis	11	23	-1.62	0.106176
Full Water Leasing	14	20	-1.34	0.179571
Wet Meadow	15	19	-0.95	0.342694

The preceding results would appear to mean that the mixing in Glendo Reservoir would negate the TDS decreases observed at the Orin gage. However, this is not exactly true. The annual mean TDS data for Glendo Reservoir for each of the Program alternatives and the Present Condition are shown on Figure 18. The *y*-axis of the time series plots on the left of Figure 18 range from 350 to 550 mg/L, which is a lower range of TDS than was shown on Figure 17. What has be negated is the difference in TDS between the Program alternatives and the Present Condition, not the effect on TDS dilution.

Figure 18 indicates that the differences between the mean annual TDS of the Program alternatives are confined to the early part of the record. In the second half of the period 1961 through 1994, there is near complete overlap between the mean annual TDS of the alternatives and the Present Condition. This pattern that was shown by the TDS of the Governance Committee and Full Water Leasing alternatives relative to the Present Condition at the Orin gage is now common to the other 2 alternatives as well. Nevertheless, there are still differences from the Present Condition that are comparable to those at the Orin gage. The maximum decrease in the mean annual TDS for the Program alternatives in relation to the Present Condition range from 16 to 22 mg/L with the largest decrease attributable to the Full Water Leasing alternative and the smallest to the Water Emphasis Alternative. Maximum increases range from the 11 mg/L of the Full Water Leasing Alternative to the 21 mg/L of the Wet Meadow Emphasis Alternative. The reductions in TDS relative to the Present Condition attributable the Full Water Leasing Alternative are an indicator that the volume of the reservoir is a factor in reducing the TDS in Glendo Reservoir. Pool volumes remain higher with the implementation of the Full Water Leasing Alternative.

As was the case in the other three reservoirs upstream, monthly changes in TDS from that of the Present Condition with the various alternatives are much larger than those of the annual changes. The maximum monthly increases and decreases, along with the net change in TDS from the Present Condition, are shown in Table 12. Actually, the range between the maximum monthly increases and decreases in the Glendo Reservoir TDS are as large or larger than those in Seminoe Reservoir. This reflects the fact that the TDS in Glendo Reservoir is controlled more by local conditions than by the upstream reservoirs. This is a further indication of the influence of the saline inflows between Alcova Reservoir and the Orin gage. However, another factor in the

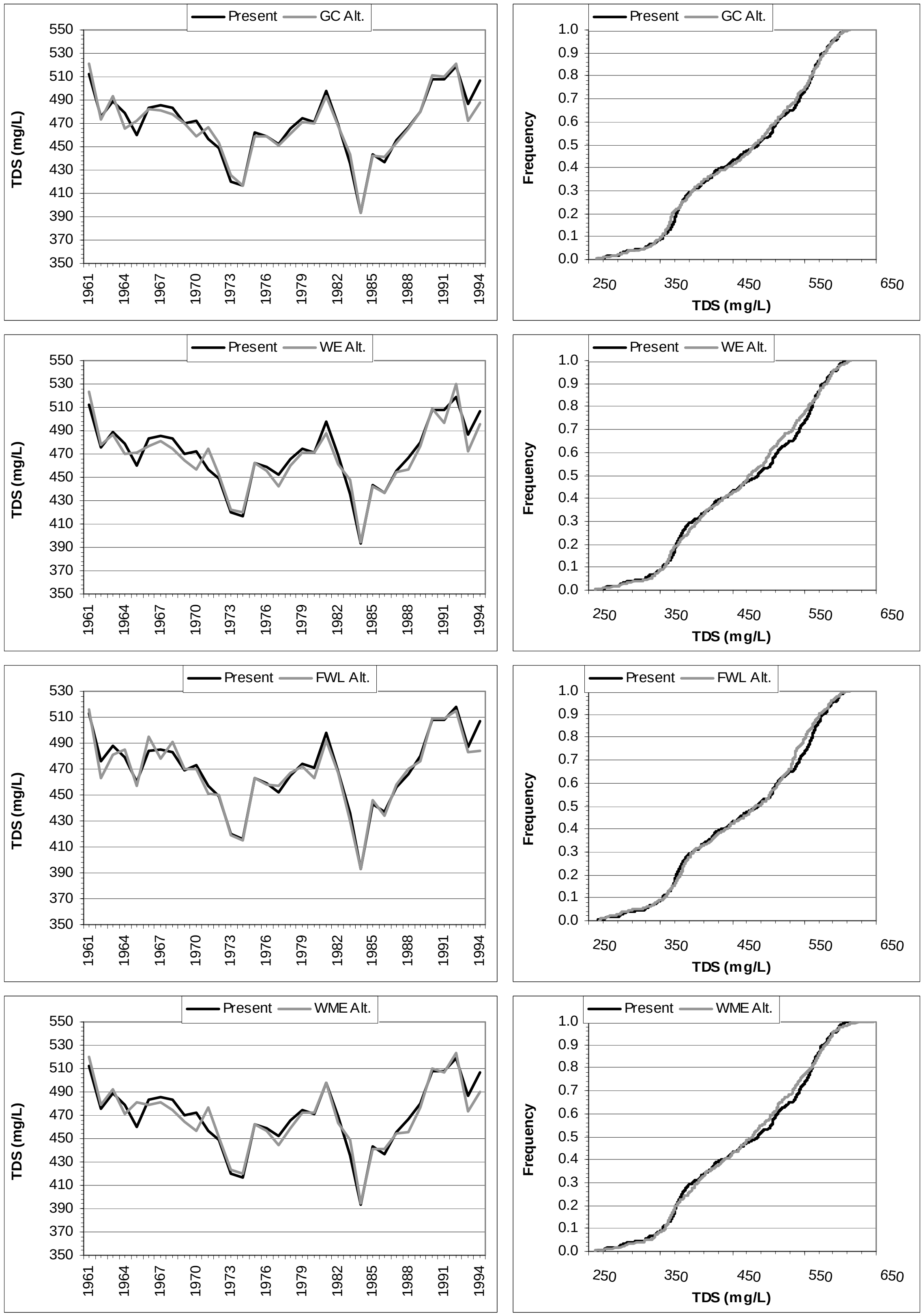


Figure 18. Comparison of the mean annual TDS and the cumulative frequency of the monthly TDS of the Present Condition and each alternative in Glendo Reservoir

response in Glendo Reservoir is that the residence time is much shorter than that of Seminoe Reservoir. On the average, the residence time in Seminoe Reservoir is about 8 months for the Present Condition and each of the alternatives, while in Glendo Reservoir, the average residence time is less than ½ that. The shorter residence time in Glendo Reservoir leads to a more rapid response of the reservoir TDS to its inflow concentrations. The net effect is a wider range in its TDS increases and decreases.

Table 12. Changes in Glendo Reservoir TDS in comparison to the Present Condition TDS (all in mg/L)

Alternative	Maximum Decrease	Net Change	Maximum Increase
Governance Committee	-101	-1	62
Water Emphasis	-116	-2	96
Full Water Leasing	-120	-2	82
Wet Meadow Emphasis	-117	-1	103

Based on the monthly data, all of the alternatives are projected to show a net decrease in TDS relative to the Present Condition (Table 12). The maximum monthly TDS increase from the Present Condition in Glendo Reservoir is shown by the Wet Meadow Emphasis Alternative (Table 12), while the maximum decrease is shown by the Full Water Leasing Alternative. In all cases, the alternatives show a larger maximum decrease, than their respective maximum increase in TDS. The Governance Committee Alternative shows the smallest maximum increase in TDS relative to the Present Condition, but it also shows the smallest maximum decrease in TDS. In other words, the Governance Committee shows the smallest range of changes in the Glendo TDS concentration from that of the Present Condition. However, the net changes are only 1 or 2 mg/L over the entire period used in the analysis. Once again, the larger changes are rather transient, and based on residence times, are more transient than in any of the other reservoirs in the system.

Temperature and DO of the North Platte River and Reservoirs

In conjunction with the preparation of the Draft Environmental Impact Statement (DEIS) for the Platte River Endangered Species Recovery Implementation Program, an analysis of the potential effects on fisheries in the Upper North Platte Basin mainstem reservoirs in Wyoming (Figure 1) was conducted. The analysis was conducted in response to concerns raised by the Wyoming Game and Fish Department (WGFD). Prior to the preparation of the DEIS, the WGFD (2004) had indicated that the impacts on the fisheries would be devastating if the contents of Seminoe or Pathfinder reservoir fell below 50,000 acre-feet or Glendo Reservoir fell below 64,000 acre-feet (critical storage). The analysis of the delivery of additional water to the Central Platte from the North Platte system using the North Platte Basin EIS Operations Model indicated that each of the reservoirs would fall below their respective critical storage level. The WGFD recommended that the potential effects be analyzed using a morphoedaphic index (MEI), coupled with regression relationships that the Department had developed for fisheries management. The WGFD felt that the results of that study would aid in developing mitigation measures for the loss of the North Platte reservoir fisheries.

A similar set of simulations were made with the North Platte Basin EIS Operations Model in conjunction with the preparation of the Final EIS (FEIS) for the Program. In the interim between the preparation of the DEIS and the FEIS, the North Platte Basin EIS Operations Model was improved to better simulate reservoir storage and North Platte Project and Kendrick Project operations. The results of the more recent operations studies do not indicate the effects on the North Platte Basin will be quite as severe as those of the operations studies for the DEIS.

Seminoe Reservoir

Table 13 summarizes the results of the operations studies in terms of minimum reservoir content and the number of times the reservoir end-of-month (EOM) content fell below 50,000 acre-feet (50 kaf). As is shown in Table 13, the overall minimum content in Seminoe Reservoir in the 48-year operations study for the Present Condition was 92 kaf. The one other alternative that did not have an EOM content below 50 kaf was the Full Water Leasing Alternative, which had an even higher EOM content than the Present Condition at about 113 kaf. The other 3 alternative each showed minimum EOM contents below the 50 kaf level, many at what is essentially the minimum pool in Seminoe Reservoir (31.2 kaf). The minimum is dictated by the elevation of the intake to the powerplant.

The minimum reservoir contents in Seminoe Reservoir for the 3 alternatives that are below 50 kaf in the months of October through April (Table 13) all occurred under the conditions represented by water year 1965. In the DEIS studies, the lowest reservoir pools occurred under the conditions represented by 1961 and 1964. This result indicates that the critical year for Seminoe Reservoir has changed for those alternatives between that in the DEIS and what is now shown for the FEIS.

The main concern over low EOM content in the DEIS was for high temperatures relative to the requirements for coldwater game fish (trout) and low dissolved oxygen (DO) when

Table 13. Comparison of end of month content (kaf[1]) among alternatives in Seminoe Reservoir													
Present Conditions	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	157.4	146.8	133.8	121.8	109.9	92.3	110.4	185.3	303.2	192.6	180.5	180	92.3
Times < 50,000 af	0	0	0	0	0	0	0	0	0	0	0	0	0
Governance Committee	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	47.7	37.6	31.2	31.2	31.2	31.2	31.2	150	276	179.6	155.3	64.1	31.2
Times < 50,000 af	1	1	1	1	1	1	1	0	0	0	0	0	7
Water Emphasis	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	37.9	31.2	31.2	31.2	31.2	31.2	31.2	147.3	265.9	179.8	131.4	54.2	31.2
Times < 50,000 af	1	1	1	1	1	1	1	0	0	0	0	0	7
Full Water Leasing	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	169.9	159.2	146.1	134.1	122.1	112.7	168	210.7	362.3	287.8	258.2	187.1	112.7
Times < 50,000 af	0	0	0	0	0	0	0	0	0	0	0	0	0
Wet Meadow Emphasis	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	31.2	31.2	31.2	31.2	31.2	31.2	31.2	118	206.1	121	31.2	31.2	31.2
Times < 50,000 af	1	1	1	1	1	1	1	0	0	0	1	2	10
[1] kaf – thousand acre-feet													

the reservoir was stratified in the summer. As can be seen in Table 13, 50 kaf critical pool level is not an issue in the summer months. Temperature is not an issue in the months in Table 13 in which the EOM content is less than 50 kaf. In addition, the reservoir is well aerated in those months in which there is not ice-cover. Generally, ice-cover is present from January through March or early April. There is usually adequate DO in the winter as well because of photosynthesis under the ice. However, if there is a thick enough snow cover, light can be cut off, eliminating oxygenation from photosynthesis. A snow cover of about 3 feet is necessary to completely cut off light penetration. This situation is considered unlikely, because of windy conditions at the reservoir, particularly during snow storms. For these reasons, the coldwater fishery should not be lost with either the Governance Committee or the Water Emphasis alternatives.

The Wet Meadow Alternative does show summer EOM contents at the minimum pool in Seminoe Reservoir on 3 occasions, once in August and twice in September (Table 13). The low August EOM content occurs under the conditions represented by 1961, while the low September EOM contents occurred the 1961 and 1965 conditions in the operations studies. If the Wet Meadow Alternative were to be implemented as the selected or preferred alternative, rather than the current preferred alternative, the Governance Committee Alternative, then further investigation may be warranted. However, no further analysis specific to Seminoe Reservoir will be attempted for the FEIS, although the potential for DO depletion will be evaluated in conjunction with a study on Pathfinder Reservoir. As it stands now, the coldwater fishery would be assumed to be lost with Wet Meadow Alternative for purposes of impact analysis in the FEIS.

Pathfinder Reservoir

Table 14 shows the minimum monthly EOM contents of Pathfinder Reservoir and the number of times in the 48-year operations study that the reservoir fell below 50 kaf. Draw down is obviously more severe in Pathfinder reservoir than was the case in Seminoe Reservoir (compare tables 13 and 14). The EOM content falls below the 50 kaf critical pool level in the operations study twice even under the Present Condition, once in September (1964) and once in March (1965). The only alternative under which the EOM content does not fall below 50 kaf is the Full Water Leasing Alternative (Table 14).

Table 14. Comparison of end of month content (kaf) among alternatives in Pathfinder Reservoir

Present Conditions	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	57.8	60	60	62.5	65.8	46.8	53.3	156.9	201.7	127.9	101.4	31.4	31.4
Times < 50,000 af	0	0	0	0	0	1	0	0	0	0	0	1	2
Governance Committee	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	52.4	54.6	47.9	38.5	31.4	31.4	55.6	127.3	183.5	79.3	31.4	31.4	31.4
Times < 50,000 af	0	0	1	1	1	1	0	0	0	0	2	2	8
Water Emphasis	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	52.4	51	38.1	31.4	31.4	31.4	55.4	124.9	176.8	119.4	31.4	31.4	31.4
Times < 50,000 af	0	0	1	1	1	2	0	0	0	0	2	2	9
Full Water Leasing	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	144.6	146.6	146.3	148.7	151.9	107.8	76.5	179	241.1	191	172	124.3	76.5
Times < 50,000 af	0	0	0	0	0	0	0	0	0	0	0	0	0
Wet Meadow Emphasis	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	31.4	31.4	31.4	31.4	31.4	31.4	55	100.2	136.8	75	31.4	31.4	31.4
Times < 50,000 af	1	1	1	1	1	1	0	0	0	0	2	3	11

Each of the other 3 alternatives, the Governance Committee Alternative, Water Emphasis Alternative, and Wet Meadow Alternative, have at least 1 occasion when the EOM content falls below 50 kaf in the fall though spring months. In the case of the Water Emphasis Alternative, the EOM content falls below 50 kaf twice in March (Table 14). For the same reasons as noted above for Seminoe Reservoir, these occasions should not result in water quality conditions that would cause the loss of the coldwater fishery. However, there are at least 2 years under each of these other 3 alternatives in which the EOM content falls below the critical 50 kaf pool level in August and September. The August draw down occurs in 1961 and 1964 in all cases. The low EOM content also occurs in these years, plus 1 additional year, 1963, for the Wet Meadow Alternative. These results indicate that Pathfinder Reservoir could be subject to the most severe adverse effects in the North Platte reservoir system, and Pathfinder Reservoir is the focus of much of the effort on the North Platte Basin. Because the August and September low EOM content could result in adverse water quality conditions for coldwater fish, additional water quality studies were conducted on Pathfinder Reservoir. To evaluate the effects on temperature, a temperature model was constructed and used to evaluate the 1961 and 1964 operations. The probability of a late summer anoxic hypolimnion was evaluated using empirical models presented in Reckhow and Chapra (1983). The results of those studies are presented later in this appendix.

Alcova Reservoir

WGFD (2004) also expressed concern over potential adverse effects in Alcova Reservoir, which serves as a large afterbay for the Fremont Canyon Powerplant. The Fremont Canyon Powerplant, which generates power at the upper end of Alcova Reservoir, receives flows directly from Pathfinder Reservoir through a tunnel. The operations of Alcova Reservoir are dictated by Project water deliveries. The reservoir is maintained at full pool during the summer to provide head on the Casper-Alcova Canal, which delivers water by gravity to Kendrick Project lands near Casper. The reservoir is drawn down by 10 feet in the winter to prevent ice damage to the canal headworks. This operation would not change with any of the alternatives, and no adverse effects to Alcova Reservoir are anticipated and nor further analysis of potential impacts was undertaken.

Glendo Reservoir

Glendo Reservoir is considerably shallower than either Seminoe or Pathfinder reservoirs. It also sits considerably lower in the basin than either those reservoirs (Figure 1). As such, its inflow in late summer tends to be somewhat warmer. Because of all of these considerations, the critical pool is larger at 64 kaf. Table 15 shows a breakdown of the minimum EOM contents and the number of times the reservoir fell below 64 kaf with each of the alternatives and the Present Condition. Table 15 indicate that the reservoir level could fall below 64 kaf in EOM content in between 1 and 9 years in September, depending on the alternative. The single occurrence was under the Present Condition and occurred under the conditions represented by 1964 in the operations study. Even the Full Water Leasing Alternative showed a number of times that the EOM content fell below the 64 kaf critical pool level, all of which occurred in the 1960s (1961, 1964, and 1966). The Governance Committee Alternative shows 1 additional year in which the EOM content fell below 64 kaf in Glendo Reservoir, 1963. The remaining 2 alternatives show 8 or 9 years in which the September EOM content fell below 64 kaf. The additional years were mainly in the 1960s for the Water Emphasis Alternative (1960, 1967, and 1969), but also included 1992. The Wet Meadow Alternative generally included that same years as the Water Emphasis Alternative, but 1955 and 1959 were substituted for 1961.

As is shown in Table 15, all of the EOM contents that are below 64 kaf are less than 1 kaf below that critical pool level at 63.1 kaf. A further analysis of the operations model indicated that the result was probably not different from the 64 kaf critical EOM content. It was generally agreed between the EIS team and WGFD representatives that, although coldwater fish would probably be stressed, the fishery was not likely to be lost.

To further refine the projected water quality conditions that could be anticipated if Glendo Reservoir fell below 64 kaf in September, historic water quality data from the reservoir were reviewed. The USGS monitored the water quality in the North Platte reservoirs, including Glendo Reservoir, in the 1970s. For several years, periodic profiles of temperature and DO, among other water quality constituents, were measured (or sampled) at 2 sites in Glendo Reservoir. Monthly temperature and DO profiles for the period August through November 1974 are shown on Figure 19. When the September profile was measured, the reservoir content was

Table 15. Comparison of end of month (kaf) content among alternatives at Glendo Reservoir													
Present Conditions	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	101.5	136.8	167.5	200.5	235.8	278.3	285.6	292	219.2	210.1	80	63.1	63.1
Times < 64,000 af	0	0	0	0	0	0	0	0	0	0	0	1	1
Governance Committee	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	91.6	126.4	155.8	186	220	253.3	249.9	285.5	275.2	198.3	80	63.1	63.1
Times < 64,000 af	0	0	0	0	0	0	0	0	0	0	0	4	4
Water Emphasis	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	91.6	126.4	155.8	186	220	253.3	243.1	281.5	275.3	146.9	80	63.1	63.1
Times < 64,000 af	0	0	0	0	0	0	0	0	0	0	0	8	8
Full Water Leasing	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	91.6	126.6	158.2	201.6	237.6	283.6	290.9	252.5	300.9	241.3	80	63.1	63.1
Times < 64,000 af	0	0	0	0	0	0	0	0	0	0	0	3	3
Wet Meadow Emphasis	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Overall
Minimum	91.6	126.4	155.8	186	220	252	243.1	284.9	271.7	90	80	63.1	63.1
Times < 64,000 af	0	0	0	0	0	0	0	0	0	0	0	9	9

approximately 63 kaf. The series of profiles on Figure 19 should give an indication of the conditions that can be anticipated when the reservoir is drawn down.

The set of plots on Figure 19 show temperature and dissolved oxygen profiles in Glendo Reservoir in the month preceding drawdown to less than 64,000 acre-feet of storage and the 2 months following at 2 different sites. Site 1 (S1 on the plots) is located about 150 feet up the reservoir from the dam; site 2 (S2) is located approximately 2.8 miles up the reservoir from the dam.

At the end of August, when the reservoir pool level was still in excess of 100,000 acre-feet, the reservoir showed a significant deepening of the epilimnion. The temperature at site 1 was 19.5°C (67.1°F) in the upper mixed layer, which extended to a depth of about 50 feet. Coincidentally, the dissolved oxygen (DO) was at or above 6 mg/L. Conditions were similar at site 2, but the depth of the mixed layer was only about 30 feet. The conditions reflect instability with respect to thermal stratification and indicate that fall overturn was already in its early stages.

The reservoir content at Glendo was 62,720 acre-feet of water at the time the measurements were made in September 1974. There were sharp gradients of both temperature and DO at the time. The temperature was suitable for salmonids throughout the profile, but adequate DO was only present in the upper 20 to 30 feet of water. The deeper layer of suitable DO was at site 2.

Ammonia, including unionized ammonia or NH_3 and the ammonium ion or NH_4^{+1}, is highly toxic to fish (EPA, 1985; 1999). Ammonia is formed during nitrification and denitrification, and is more stable under somewhat reducing conditions, but is not an endpoint of either mechanism. NH_3 in water equilibrates against NH_4^{+1}. The equilibrium is temperature and pH dependent. The acute ammonia criteria are pH dependent, while the chronic criteria are pH and temperature dependent (Attachment A). There are no pH data for Glendo Reservoir at the time the temperature and DO profiles were measured, but the pH of the river downstream from the dam

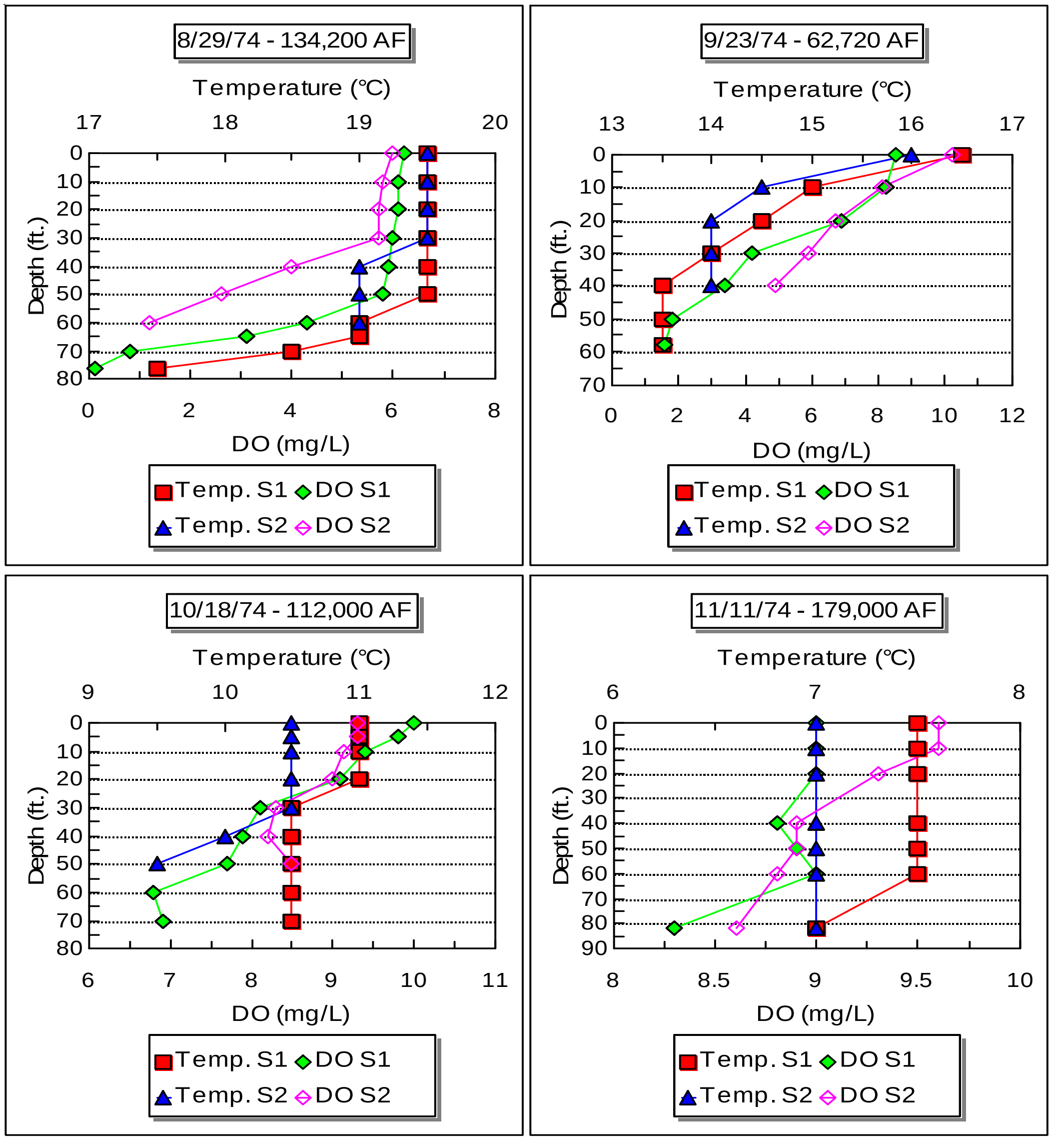


Figure 19. Late summer and early fall temperature and dissolved oxygen profiles in Glendo Reservoir in 1974 (USGS, 1975; 1976)

was 8.3; the chronic total ammonia concentration of the criterion at that pH and the temperature of the reservoir would be about 1.6 mg/L.

Total ammonia concentrations at site 1 on September 23, 1974 ranged from 0.13 mg/L at the surface to 0.29 mg/L at the bottom of the profile. Ammonia concentrations were lower (0.22 mg/L) at site 2, where the DO was somewhat higher. Total ammonia should not have been at a toxic concentration in September 1974. However, the minimum DO in the 1974 profiles occurred in August (Figure 19). At that time, the total ammonia concentration in the bottom of Glendo Reservoir was 0.43 mg/L at site 1 and 0.38 at site 2. There was no pH measurement in the river downstream at the time, but the pH at the downstream river site was 8.3 on all dates from July through September 1974. Assuming the bottom pH in the reservoir was also 8.3 mg/L in August yields a total ammonia criterion of 1.6 mg/L. As an aside, the minimum total ammonia criterion in EPA (1999) when early life stages are absent at any temperature and pH is 0.44 mg/L (Attachment A), which is approached in August, but not exceeded.

As is shown on Figure 19, the reservoir had increased to over 100,000 acre-feet by the time of the October sample. The reservoir continued to cool and was becoming more oxygenated in all of the profiles in October and November. The conditions in October indicate that there was a relatively rapid recovery of water quality in the reservoir.

North Platte River

WGFD (2004) also expressed concern over the possible effects of excessive temperature and low DO in reservoir releases on several downstream river reaches. The river reaches include the river between Kortes Dam and Pathfinder Reservoir, a.k.a., the "Miracle Mile", the newly established (2002) Cardwell fishery in Fremont Canyon, and the river downstream from Gray Reef Dam, which forms an afterbay for the Alcova Powerplant.

Seminoe and Kortes Dams and the Miracle Mile

Kortes Reservoir is a small afterbay that is used for flow equalization for peaking power generation releases from Seminoe Powerplant. The average residence time in Kortes Reservoir is only a little over 1.5 days. As a result, the water in Kortes Reservoir reflects the quality of the releases from Seminoe Dam. This is illustrated by the temperature and DO profiles in Attachment B. Attachment B consists of a series of temperature and DO profiles measured in the 2 reservoirs, either on the same day or on subsequent days, but never more than 1 day apart. The outlet elevation is flagged on the Seminoe Reservoir profiles. The temperature and DO at the Seminoe Reservoir outlet elevation should reflect the approximate temperature and DO concentration of the release to Kortes Reservoir. The profiles are annotated to indicate similarities and differences between the estimated quality of the Seminoe Dam releases and the profiles in Kortes Reservoir. The profiles are taken from the field notes collected by Sartoris *et al.* (1981).

Sartoris *et al.* (1981) also measured the temperature and DO concentrations of the Kortes Dam release and in the North Platte River in the Miracle Mile. However, the Kortes Dam measurements were made in 1977, while the measurements in the Miracle Mile were made in 1978 and 1979. The fact that the data are from different years makes it difficult to draw inferences because of interannual variation in weather.

In 1982 and 1983 in a later study of Seminoe Reservoir related to the potential for developing a pump-back storage powerplant, Grabowski and Sartoris (1984) measured the temperature and DO concentrations of the releases from Seminoe and Kortes dams and the river at a site located in the Miracle Mile. The data were only collected on 5 dates, but they indicate some interesting relationships.

Figure 20 shows a scatterplot of the 1982/83 temperature data, along with a regression relationship between the temperature data from the adjacent sites. The plot on the left shows the relationship between the temperature of the Seminoe Dam release and the equivalent temperature at Kortes Dam. The slope of the regression line is essentially 1. The r^2 of the regression is also essentially 1. Both of these factors indicate that the temperatures are directly related. The intercept is relatively small, but it does indicate that there is a slight bit of warming at lower release temperatures, and a bit less warming at higher temperatures. These relationships are based on the fact that the slope of the line is actually very slightly less than 1 (*i.e.* 0.99).

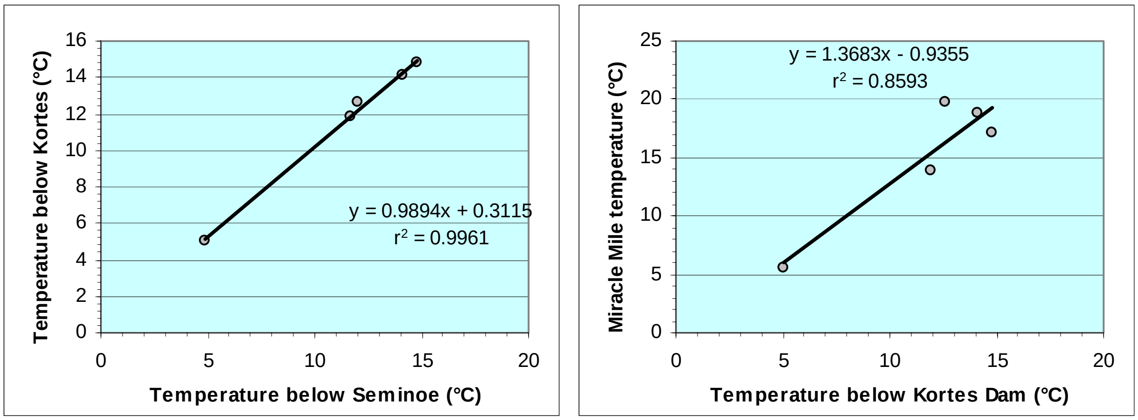


Figure 20. Interrelationships among the Seminoe and Kortes release temperatures and the temperature in the Miracle Mile (Grabowski and Sartoris, 1984)

The plot on the right of Figure 20 shows the relationship between the temperature of the Kortes Dam release and that in the Miracle Mile. The Miracle Mile site, according to the field notes, is located near the county road and about ¼ mile downstream from the USGS stream flow gage. The USGS gage is in turn located ~⅓ mile downstream from Kortes Dam. So the location of the Miracle Mile site shown on Figure 20 would be about 0.6 mile downstream from Kortes Dam. There is considerably more scatter in the Kortes–Miracle Mile temperature data than was the case with the Seminoe–Kortes temperature data, but the regression is still reasonably good, with an r^2 of about 0.86 (Figure 20). The relationship between the temperature in the Miracle Mile and that in the Kortes Dam releases indicates there is some warming at higher temperature, but the temperature in the Miracle Mile is directly related to the Kortes release, and thus to the Seminoe release.

Figure 21 shows a similar set of plots to Figure 20 for the DO data from the Seminoe and Kortes releases and the Miracle Mile. As was the case with temperature, the DO in the Seminoe Dam release shows a very good relationship to the DO in the Kortes Dam release (Figure 21). The slope of the regression line has a slope near, but slightly less than 1. However, the intercept is near 1, indicating the addition of nearly 1 mg/l of DO between Seminoe Dam and Kortes Dam. Given the effect of the slope, the addition of the nearly 1 mg/L is most effective at lower DO concentrations. This makes sense in that the higher concentrations are near saturation and DO would be physically difficult to add in the absence of turbulence. Alternatively, based on the right-hand scatterplot on Figure 21, there is no relationship between the DO in the Kortes Dam release and the DO in the Miracle Mile. The DO at the Miracle Mile site ranges from a low of 9 mg/L to a high of 13.8 mg/L, independent of the DO at Kortes Dam.

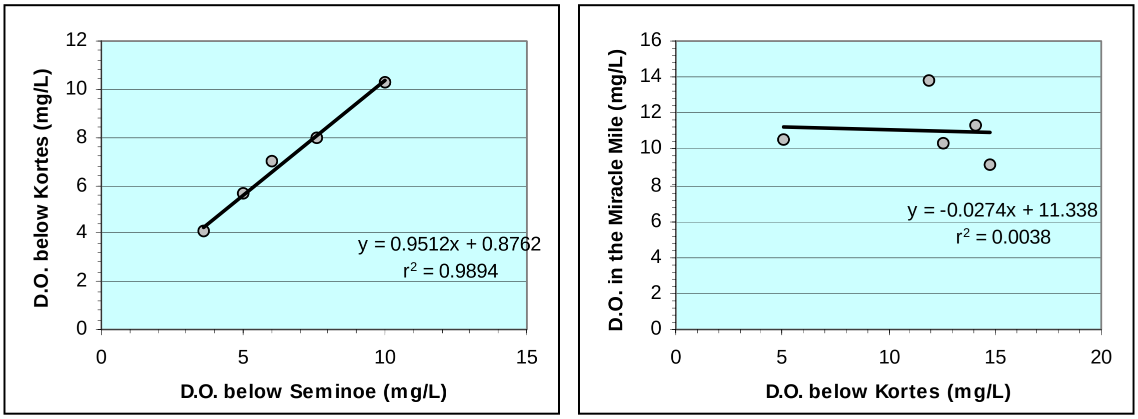


Figure 21. Interrelationships among the Seminoe and Kortes release DO and the DO in the Miracle Mile (Grabowski and Sartoris, 1984)

The relationship to DO saturation at the 3 sites is shown on Figure 22. At both Seminoe and Kortes dams, the DO is at or below saturation on all of the sampling dates. The DO is only at saturation in April 1982. During April, Seminoe Reservoir would be expected to be fully mixed at a time shortly after ice-off. When the reservoir is fully mixed, DO would be expected to be near saturation, as it is on Figure 22. On all of the sampling dates, the DO at Kortes Dam is slightly more saturated than that in the Seminoe Dam release. There is a short reach of river between Seminoe Dam and Kortes Reservoir that apparently allows for some aeration between Seminoe Dam and Kortes Reservoir. Alternatively, the DO at the Miracle Mile site is consistently above saturation, actually well above saturation on most sample dates in 1982 and 1983. There is obviously considerable aeration of the flows between Kortes Dam and the sample site that is nearly 0.6 mile downstream. The aeration between the 2 sites accounts for the lack of a relationship between the DO at Kortes Dam and that at the sample site in the Miracle Mile.

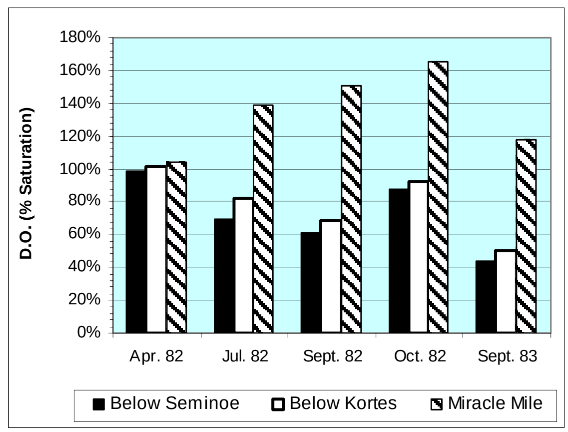


Figure 22. Percent DO saturation at the 3 sites below Seminoe and Kortes dams and in the Miracle Mile (Grabowski and Sartoris, 1984)

The USGS monitored temperature and DO at the gage site in the Miracle Mile from 1969 through 1982 and then, following a 5 year hiatus, in water years 1988 and 1989. The USGS data from 1969 through 1982 are plotted on Figure 23. The USGS data encompass a much longer period and are much more extensive than the data of Grabowski and Sartoris (1984). However, the maximum temperature in the USGS Miracle Mile data set is 16°C, while in the Grabowski and Sartoris (1984) data set, the maximum is 19.7°C or nearly 4°C higher. This would indicate that there may be some warming between the 2 sample sites, which, as was noted above, are about ¼-mile apart.

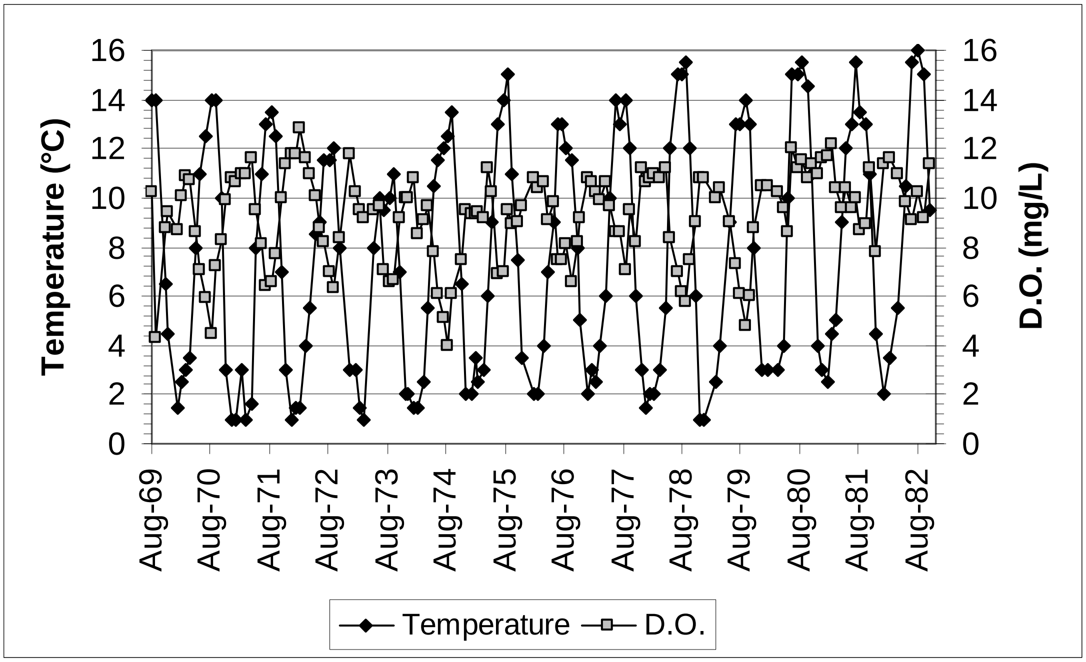


Figure 23. USGS temperature and DO measurements in the Miracle Mile from 1969 through 1982

Another interesting observation in the USGS data set concerns the minimum DO concentration. The overall minimum DO is 4 mg/L (Figure 23). There are 2 other DO concentrations between 4 mg/L and 5 mg/L. All of these low DO observations occurred in August and September when the bottom DO in Seminoe Reservoir would have been minimal. The low DO in the Miracle Mile would likely reflect that in the Seminoe Dm releases and would indicate that reaeration at the USGS gage site was not quite as effective as at the sampling site of Grabowski and Sartoris (1984) farther downstream.

Table 13 indicated that there would be no severe drawdown of Seminoe Reservoir in the summer months due to any of the alternatives being considered in the Platte River Endangered Species Restoration-Implementation Program. As a consequence, the reservoir should not warm much more than has been the case historically. On this basis, the temperature in the Miracle Mile should not be affected greatly either. For this reason additional, more intensive studies of the effects of the Program on Seminoe Reservoir are not considered necessary as part of the programmatic FEIS.

Pathfinder Dam and the Cardwell Fishery

As was noted above, the Cardwell fishery in Fremont Canyon below Pathfinder Dam was established in 2002 with the provision of a 75 ft^3/s turbine bypass flow at the Fremont Tunnel between Pathfinder Reservoir and Fremont Canyon Powerplant. There are no data available to attempt the type of evaluation of the relationship between the Cardwell fishery and Pathfinder Reservoir that was done above for the Miracle Mile.

As is shown in Table 14, Pathfinder Reservoir will be drawn down below the WGFD critical pool level of 50 kaf in both August and September in 2 years out of the 48-year study period of the North Platte Basin EIS Operations Model. The effects of the draw down are unknown. In an attempt to evaluate what would occur if the reservoir were drawn down to what is essentially its minimum operating level, a mathematical temperature model of the reservoir was constructed based on the most severe drought year for which there were data available to do so. The model construction, calibration, and results are reported elsewhere in this Appendix. In addition to the temperature model, empirical models presented in Reckhow and Chapra (1983) were used to evaluate the potential for late summer anoxia in Pathfinder Reservoir. The derivation and application of those models will also be presented later in this report. The estimated Pathfinder Dam outlet water quality from these models will be applied to the Cardwell fishery.

Alcova and Gray Reef Dams and the North Platte River Downstream

In conjunction with the reservoir studies, Sartoris *et al.* (1981) measured temperature and DO downstream from Alcova Dam in the river between the dam and Gray Reef Reservoir. There were 2 measurements each in 1978 and 1979. No measurements were made downstream from Gray Reef Dam. The data from below Alcova Dam are shown on Figure 24, which shows temperature, DO, and the percent DO saturation.

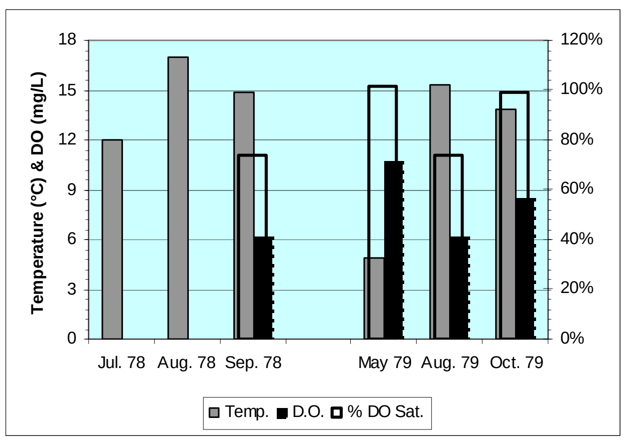


Figure 24. Temperature and DO below Alcova Dam in 1978 and 1979 (Sartoris *et al.*, 1981)

As can be seen on Figure 24, the maximum temperature below Alcova Dam occurred in August 1978, but was less than 18°C. The temperature in August 1979 was somewhat lower at about 15°C and about what had been observed in September 1978. In all cases, the release temperature is well within the tolerance of coldwater fish.

There are only 4 DO measurements shown on Figure 24, none of which coincide with the maximum release temperature. The minimum DO was 6 mg/L, which was measured in both September 1978 and August 1979. In both cases, the 6 mg/L of DO was about 75 percent of saturation. The other 2 DO measurements were at approximately 100 percent of saturation.

Alcova Reservoir, among other things, is an afterbay for Fremont Canyon Powerplant. The flow through Fremont Canyon Powerplant is conveyed through Fremont Tunnel from the intake at Pathfinder Reservoir. A set of annotated temperature and DO profiles from Pathfinder and Alcova reservoirs, similar to those of Seminoe and Kortes reservoirs previously shown in Attachment B, is included in Attachment C. The main difference in the relationship shown between Seminoe and Kortes reservoirs and that shown between Pathfinder and Alcova reservoirs is that Alcova Reservoir is much larger than Kortes Reservoir. Another important difference is between Kortes and Alcova dam. The intake to the Alcova Powerplant is near the surface – at a depth of only 20 feet during the summer. At this depth, the released water would be expected to be somewhat warm. However, in one of those unforeseen quirks of good fortune (and physics), the level of the outlet appears to be greatly responsible for dictating the level of the thermocline in the reservoir. Because of the level of the thermocline, which is very near the level of the outlet, withdrawals for release to the river come from within the thermocline. Consequently the release temperatures are below what would ordinarily be expected from a near surface withdrawal (review the profiles in Attachment C). It appears that this interrelationship between the level of the intake to the powerplant, and thus the river outlet, keeps the temperature in a suitable range for trout in the river downstream from the dam.

A further review of the profiles in Attachment C indicates a minimum DO of around 6 mg/L in the vicinity of the Alcova Dam outlet. This may be coincidence (and it probably is), but 6 mg/L is the minimum measured DO on Figure 24. In an attempt to investigate this a bit further, the DO concentration was estimated from a simplified calculation from the profiles in Alcova Reservoir. The DO concentration was taken as the average of DO concentrations in the profile within the upper and lower elevations of the intake to the outlet. No attempt was made to calculate a withdrawal zone, which would probably extend beyond those limits on most occasions. The results, along with a calculated DO saturation concentration based on the temperatures associated with the DO profiles (adjusted for the elevation of Alcova Reservoir), are shown on Figure 25. Figure 25 indicates that the DO concentration is generally above saturation, but falls below it at around 6 mg/L in August or September of some years. The data plotted on Figure 25 provide some support for all of the preceding hypothesizing, but the data are questionable in terms of the actual outlet DO. They should be considered "ballpark estimates."

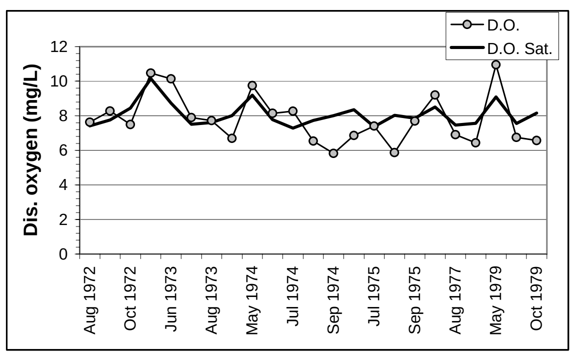


Figure 25. Estimated dissolved oxygen and saturation concentration in the releases from Alcova Dam

The USGS gage below Alcova Dam is actually located 0.8 mile below Gray Reef Dam. The USGS monitored temperature at that gage from 1970 through 1996 and DO from 1974 through 1996. This is a much longer record and a much better data set to characterize the temperature and DO below Gray Reef Dam in terms of trout habitat support than the above described Reclamation data set.

A time series plot of the USGS temperature data for the summer months (June through September) for the period 1970 through 1996 appear on Figure 26. The maximum temperature on Figure 26 is 20°C (68°F), which is the upper limit for the support of a cold water fishery, based on WGFD standards. Even if there is some warming in Gray Reef Reservoir, which is shallow, the warming is not great enough to increase whatever the release temperature from Alcova Dam might be to a temperature greater than 20°C. The potential saving grace in all of this is that in the area of the reservoir, natural water begins to cool in August, rather than heat up, at least in most years. The fact that the mass of the reservoir maintains heat later in the year would be a factor in warming the river later in the year, potentially contributing to the productivity of the fishery.

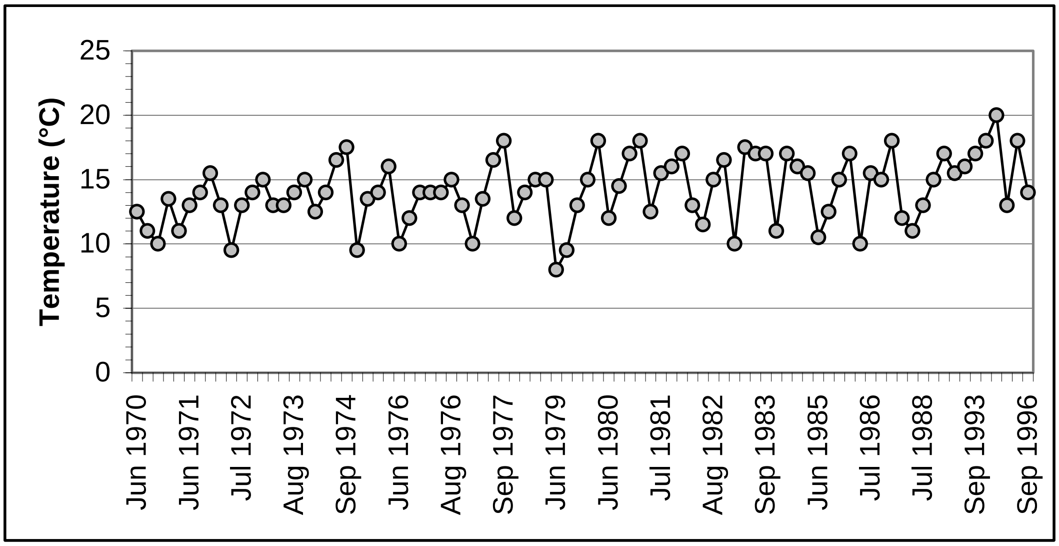


Figure 26. Summer temperatures downstream (0.8 mile) from Gray Reef Dam – with 1 exception, always < 20°C (68°F)

Figure 27 shows a plot of the 1970 through 1996 summer DO data. Also shown on the plot, as was done on Figure 25, are the DO saturation concentrations. With few exceptions, the DO concentration in the river is at or above saturation. The minimum DO concentration on Figure 27, while less than the saturation concentration, is slightly above referenced 6 mg/L benchmark. The fact that all but 4 observations are at or above saturation indicates that there is some, and probably considerable, aeration between Alcova Dam and the Alcova gage.

One interesting observation that may be a factor in the aeration downstream from Alcova Dam is the fact that there is virtually always water running down the spillway in the summer. Because the reservoir trends from west to east and is maintained full, wind-driven wave action washes water over the spillway into the river below. This overflow on the spillway could easily provide an addition of well aerated water to supplement the powerplant releases and in the process help to aerate the river downstream.

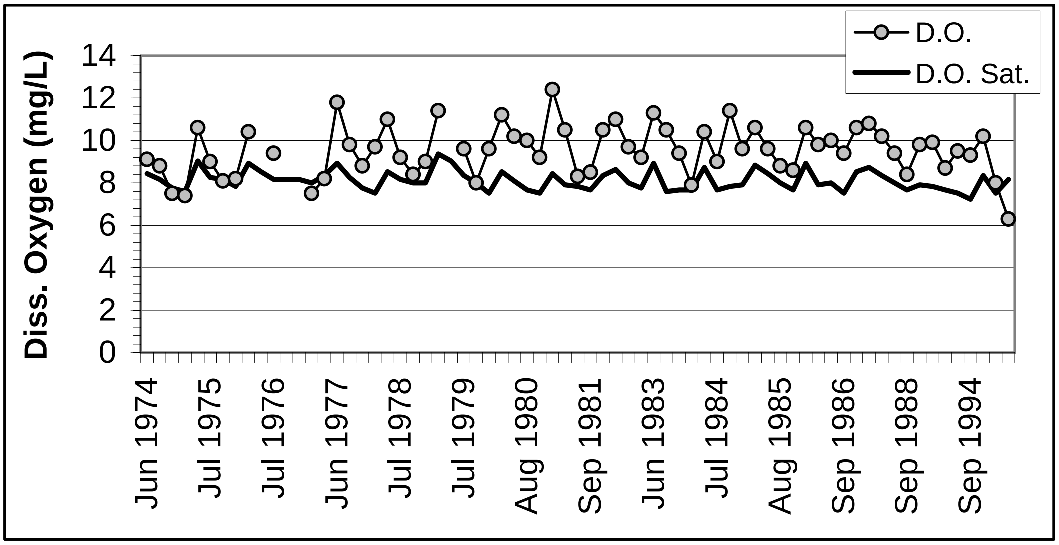


Figure 27. Summer dissolved oxygen and saturation concentrations downstream (0.8 mile) from Gray Reef Dam

There is unlikely to be any effect on temperature and DO below Gray Reef Dam due to any of the Program alternatives. First, there will be no effect on the operating levels of Alcova Reservoir, through which virtually all of the water flowing out of Gray Reef Reservoir must pass. Second, the near surface release from Alcova Reservoir is within the temperature and DO requirements for trout in the summer. Third, the temperature and DO of the flows below Gray Reef Dam have historically been within the tolerance of coldwater fish and none of the alternatives should do anything that would change that. For these reasons, at least from a water quality perspective, there should be no effect on the coldwater fishery downstream from Gray Reef Dam due to any of the Program alternatives.

Pathfinder Temperature Model

Table 14 shows that the end-of -month content Pathfinder Reservoir would fall below 50,000 acre-feet more frequently than under current conditions under 3 of the 4 Program alternatives, including the Governance Committee, Water Emphasis, and Wet Meadow Emphasis alternatives. The WGFD (2004) established the 50,000 acre-foot pool as being critical for maintaining the trout fishery in Pathfinder Reservoir. Because existing information was inadequate to define conditions with the various alternatives, a study was designed to do so. The study was to consist of two parts, a temperature model of Pathfinder Reservoir, and application of empirical nutrient loading models to estimate the probability of the late summer hypolimnion becoming anoxic. This section of the report will describe the temperature model and its results.

Methods

A mathematical temperature model of Pathfinder was constructed using the U.S. Army Corps of Engineers reservoir water quality code, CE-QUAL-R1 (WES, 1995). The code is developed such that the thermal portion of the model (CE-THERM-R1) can be run separately from the full water quality analysis model. The model was downloaded from the WES FTP site as 4 “zipped” files. The extracted files included executable programs, along with the FORTRAN source code for both CE-QUAL-R1 and CE-THERM-R1. The executable programs had been compiled with a Lahey 32-bit FORTRAN compiler under the Windows 95/98 operating system and failed to run under the current Windows XP operating system in use in Reclamation’s TSC (Technical Service Center). Inquiries to WES indicated that similar problems had been encountered in switching to certain other computers and operating systems, but recompiling the program usually solved the problem. A surplus Digital Visual FORTRAN 32-bit compiler was obtained and installed on the Windows XP computer that was to be used to run the model. The CE-THERM-R1 portion of the program was recompiled. The program subsequently ran with no problems.

The original package downloaded from WES included pre- and post-processors. These were DOS windows programs that would not function in XP and could not be used. The CE-THERM-R1 program was run in a DOS window. Pre- and post-processing of the model inputs and outputs were accomplished in an Excel spreadsheet.

According to WES (1995), CE-THERM-R1 is spatially one-dimensional and horizontally averaged. Temperature and concentration gradients are computed only in the vertical direction. The reservoir is mathematically represented as a vertical sequence of horizontal layers of variable thickness, where thermal energy and materials are uniformly distributed in each layer.

Data requirements and data set development

The CE-THERM-R1 model requires hydrologic, water quality, and weather data as drivers. Historic flow and water quality data were downloaded from the USGS (U.S. Geological Survey) NWIS (National Water Information System) website. The major tributaries to Pathfinder Reservoir include the North Platte and Sweetwater rivers. There are gages with water quality records on both rivers upstream from Pathfinder Reservoir.

The model requires 5 weather measurements. All of the measurements are usually only available from Class A weather stations. The nearest Class A weather station to Pathfinder Reservoir is at the Casper-Natrona County International Airport (Figure 1). Weather data for the Casper-Natrona County International Airport were purchased from the National Climatic Data Center for the period 1970 through 2004.

In addition to the hydrologic and weather data needed to drive the model, reservoir data are needed to calibrate the model. For a temperature model, only temperature profiles of the reservoir are needed. However, CE-THERM-R1 also simulates total dissolved solids (TDS) and total suspended solids (TSS), which can be used to evaluate mixing within the profile. Measured profiles from Pathfinder Reservoir are available from Reclamation studies conducted from 1977 through 1979 (Sartoris *et al.*, 1981) and USGS monitoring data from 1973 through 1977. The Reclamation data were not included in the study report, but the original data books are still on file. The original field data books were obtained from Jim Sartoris, the principal author of a Reclamation report on the limnology of the Upper North Platte Reservoirs (Sartoris *et al.* 1981), and the data were entered into an Excel spreadsheet for use in this study. The USGS data were not available through NWIS; so the published profiles were also entered into an Excel spreadsheet from the Annual Water Data Reports for Water Years 1973 through 1978, except for those for Water Year 1976. The profiles for Water Year 1976 were not published, although a note in the Water Data Report for that year indicates that the data are on file in the USGS office in Casper.

The primary reason for modeling Pathfinder Reservoir was to evaluate conditions during extreme drawdown. Such drawdown occurs during drought conditions. In the North Platte EIS Operations Study, Pathfinder Reservoir was drawn down to minimum pool during the simulated years 1961 and 1964. There were insufficient data for those years to conduct a direct reservoir water quality study. Of the years for which measured profiles are available, the greatest drawdown occurred during 1977, when the reservoir content was reduced to 249,000 acre-feet on September 30. Although that was still well above minimum pool, it was the lowest content at which a calibration could be evaluated. In addition to a low pool level, there were more profiles available from 1977 than from any other year. Consequently, 1977 was selected for the initial simulation. Data sets for 1973 and 1974 were pulled together, but were not used in the study. Those data sets would be available for additional study. However, both 1973 and 1974 were above average water years and would not be likely to provide a better extrapolation to low water conditions than the 1977 data set.

Weather Data

The 1977 weather data obtained from the Casper-Natrona County Airport weather station were incomplete in terms of the model requirements. The CE-THERM-R1 model needs measured air temperature, dew point temperature, station pressure, wind speed, and cloud (sky) cover. Station pressure is the barometric pressure uncorrected to sea level, the latter of which is the usually published measurement. Of the required data, only air temperature and sky cover were available for 1977. All of the required measurements were only available beginning in 1984. To fill in the gaps, relationships among the measurements were explored.

On the basis of the assumption that similar water years would have similar weather patterns, the mean annual flows for the period that weather data were available (1970-2004) were compared. The mean annual flow for 1977 was 483 ft^3/s, which was second only to 2002 in terms of low annual flow (270 ft^3/s). The nearest low flow year to 1977 was the 3rd ranking year, 1992, which had a mean annual flow of 543 ft^3/s.

The available weather data for 1977 were compared with those for 1992 using correlation analysis. The results are shown in Table 16. There are relatively good correlations among the various temperature measurements for 1977 and 1992. The 1977 data included only the minimum and maximum daily temperatures. The average temperature shown in Table 13 for 1977 represents the average of those 2 data points. Alternatively, the 1992 data included a mean temperature calculated from the individual temperature measurements made throughout the day; this is labeled the mean temperature. The 1992 average temperature was calculated from the daily minimum and maximum temperature as was done with the 1977 data. The 2 measurements differ somewhat.

There is a very good correlation between the dew point temperature and the minimum daily temperature for 1992 (Table 16). Based on the correlation, a regression relationship was developed between the dew point temperature and the minimum daily temperature. The regression was applied to the minimum daily temperature for 1977 to estimate the dew point temperature.

The remaining 2 measurements needed for the model, station pressure and wind speed, show a number of significant correlations in Table 16. However, the correlation coefficients (r) for those correlations are relatively small and would not provide a good basis for developing regression relationships that could be transposed to another year. In the absence of a regression relationship, the 1992 data for station pressure and wind speed were directly included in the 1977 CE-THERM-R1 model.

Flow and Water Quality Data

Flow data at the USGS gage on the North Platte River upstream from Pathfinder Reservoir were only available for Water Years 1911 through 1959. The 1977 flows for that site were taken from a data set of historic flows constructed for use in the calibration of the North Platte EIS Operations Model.

The inflow to Pathfinder Reservoir from the Sweetwater River were taken from its gage near Alcova. However, in some years, including 1977, the gage was only seasonal. To fill the gaps in the Sweetwater River flow record, a regression relationship was developed with the gage on the Sweetwater River at Sweetwater Station. The missing data at the Alcova gage were from the low-flow period of the year. The Sweetwater River undergoes a net loss in flow between the 2 gages, but the loss is highly variable from year-to-year, and the regression overestimated the low-flows by quite a bit..

Table 16. Pearson correlation matrix - 1977 and 1992 weather data from the Casper-Natrona County International Airport														
Variable		Date	Temperature: Maximum – 1977	Temperature: Minimum – 1977	Temperature: Ave. – 1977	Temperature: Maximum – 1992	Temperature: Minimum – 1992	Temperature: Ave. – 1992	Temperature: Mean – 1992	Wind - 1992	Dew point - 1992	Precipitation - 1992	Pressure - 1992	Sky Cover 1992
Sky - 1977	r	-0.0573	-0.2539	-0.0797	-0.1798	-0.1447	-0.1428	-0.1482	-0.1506	0.0156	-0.1243	-0.0316	-0.0017	0.5482
	Prob > r	0.274694	0.000001	0.128770	0.000556	0.005607	0.006281	0.004562	0.003941	0.766007	0.017535	0.547487	0.973537	< 0.000001
Sky - 1992	r	-0.0544	-0.1652	-0.0841	-0.1318	-0.0704	-0.1181	-0.0936	-0.0992	-0.0738	-0.0967	-0.0310	0.0354	
	Prob > r	0.299554	0.001544	0.108537	0.011720	0.179756	0.024042	0.073983	0.058203	0.159278	0.064890	0.554664	0.500468	
Pressure - 1992	r	0.0440	0.2097	0.1993	0.2096	0.1906	0.1048	0.1587	0.1589	-0.2808	0.0539	-0.1196		
	Prob > r	0.402200	0.000054	0.000126	0.000055	0.000250	0.045425	0.002367	0.002332	< 0.000001	0.304664	0.022262		
Precipitation - 1992	r	0.0182	0.0930	0.1023	0.0993	-0.0840	0.0556	-0.0247	-0.0240	-0.0444	0.2348			
	Prob > r	0.728785	0.075982	0.050827	0.058032	0.109312	0.289720	0.637583	0.647734	0.398132	0.000006			
Dew point -1992	r	0.0386	0.7159	0.7395	0.7415	0.7474	0.8733	0.8263	0.8273	-0.3192				
	Prob > r	0.462461	< 0.000001	< 0.000001	< 0.000001	< 0.000001	< 0.000001	< 0.000001	< 0.000001	< 0.000001				
Wind - 1992	r	0.1224	-0.3485	-0.3565	-0.3598	-0.3236	-0.1797	-0.2701	-0.2718					
	Prob > r	0.019319	< 0.000001	< 0.000001	< 0.000001	< 0.000001	0.000564	< 0.000001	< 0.000001					
Temperature: Mean – 1992	r	0.0434	0.7728	0.7605	0.7833	0.9763	0.9596	0.9991						
	Prob > r	0.408493	< 0.000001	< 0.000001	< 0.000001	< 0.000001	< 0.000001	< 0.000001						
Temperature: Ave. – 1992	r	0.0446	0.7725	0.7602	0.7830	0.9775	0.9599							
	Prob > r	0.395490	< 0.000001	< 0.000001	< 0.000001	< 0.000001	< 0.000001							
Temperature: Minimum – 1992	r	0.0406	0.7377	0.7337	0.7512	0.8794								
	Prob > r	0.439763	< 0.000001	< 0.000001	< 0.000001	< 0.000001								
Temperature: Maximum – 1992	r	0.0457	0.7583	0.7402	0.7659									
	Prob > r	0.384481	< 0.000001	< 0.000001	< 0.000001									
Temperature: Ave. – 1977	r	0.1806	0.9835	0.9745										
	Prob > r	0.000525	< 0.000001	< 0.000001										
Temperature: Minimum – 1977	r	0.1642	0.9178											
	Prob > r	0.001646	< 0.000001											
Temperature: Maximum – 1977	r	0.1871												
	Prob > r	0.000326												
NOTE - Prob > r is the probability of a correlation coefficient of the magnitude shown occurring by chance alone. Number of observations: 365														

Although the flow record for the North Platte River upstream from Pathfinder Reservoir ended in 1959, water quality data for the gage are available for the period 1969 through 1989. The data include monthly measurements of water temperature and specific electrical conductance (EC). There were also measurements of air temperature later in the record. Similar data are available for the Sweetwater River at the gage near Alcova.

Initially, the water temperature data for the North Platte above Pathfinder Reservoir was developed from a regression of water temperature on air temperature. However, the North Platte River upstream from Pathfinder Reservoir is also the North Platte Reservoir downstream from Seminoe Reservoir. The temperature of the North Platte inflow to Pathfinder does not follow the seasonal pattern of the air temperature.

Figure 28 shows a plot of the mean monthly water temperatures of the North Platte and Sweetwater rivers upstream from Pathfinder Reservoir. The Sweetwater River is essentially unregulated where it enters Pathfinder Reservoir, at least as far as any influence on water temperature is concerned. Consequently, the water temperature shows a seasonal pattern similar to that of air temperature. However, where the Sweetwater shows a temperature peak in July, the North Platte does not show its maximum until August, at which time the temperature of the Sweetwater River is declining. Because the air temperature data for the North Platte River were extremely limited, a data set was developed using coincident (same day) temperature data for the North Platte and Sweetwater rivers. The water temperature data for April through October at the 2 sites are plotted against each other on figure 29.

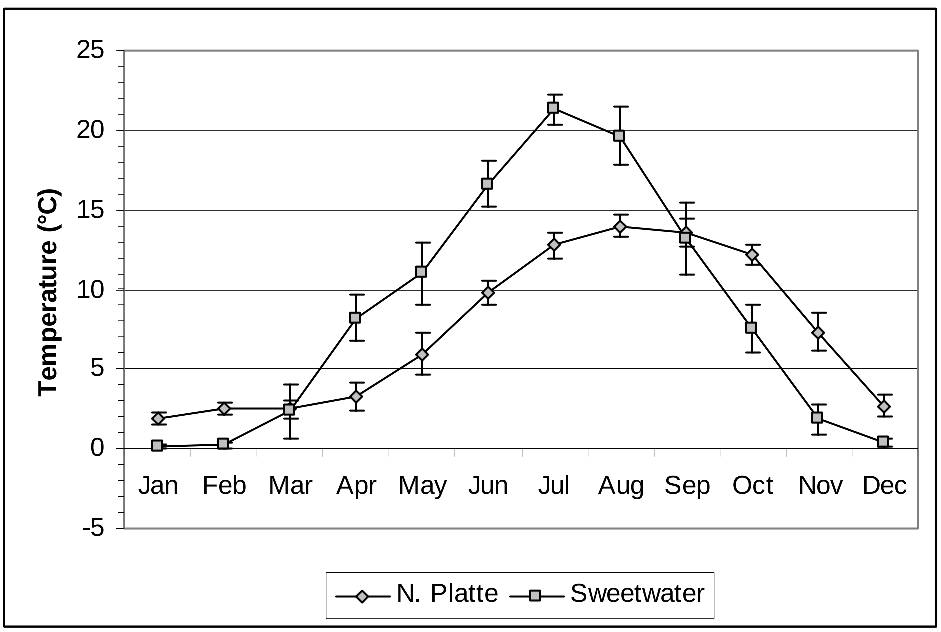


Figure 28. Monthly distribution of temperatures in the Miracle Mile and
the Sweetwater River upstream from Pathfinder Reservoir

During April through July, the temperature is rising in both rivers (Figure 29). For this period a linear regression was developed (Figure 29). This is plotted on the rising limb of the plot on Figure 29 and delineates those data on the scatter plot. During August through October, the

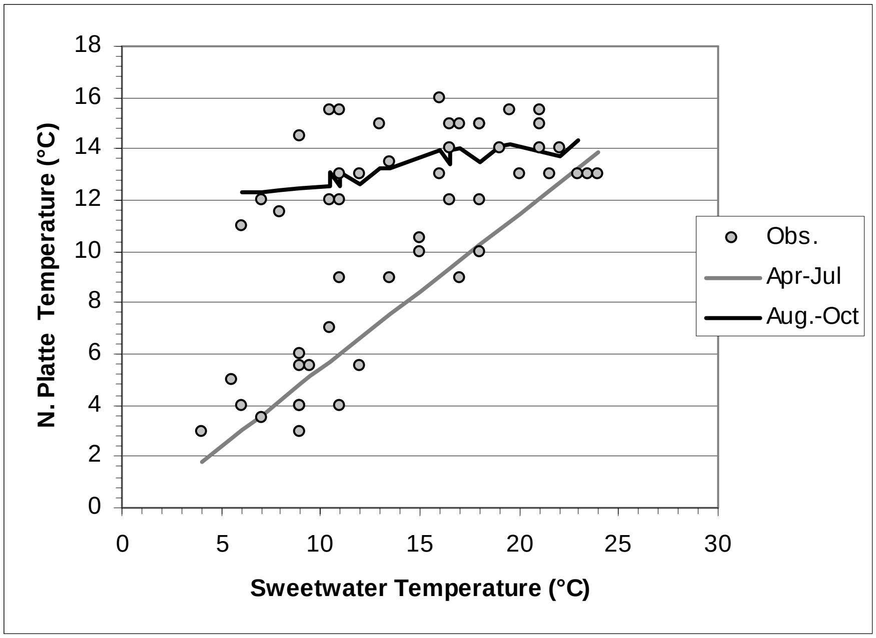


Figure 29. Relationships between the water temperature of the North Platte and Sweetwater rivers upstream from Pathfinder Reservoir

temperature of the Sweetwater River decreases, while that of the North Platte rises and falls (Figure 29), but there is only a slight change. This plots as something of a plateau on Figure 29. For this period, a multiple regression relation was developed with month as a second independent variable. The influence of month in the regression is reflected by the kinks in the August through October regression line on Figure 29. Although the multiple regression is not particularly significant, the reason for its development and use is explained below in the Calibration section.

There is usually an inverse relationship between TDS and flow in Western rivers. The relationship reflects the effect of dilution when flows are high. Just as the influence of Seminoe Reservoir affected the air temperature-water temperature, it also damps any instantaneous influence of dilution. The dilution occurs in the reservoir over a period of months during spring runoff. The variation in TDS consequently occurs more from year-to-year based on the magnitude of runoff. Rather than using a regression of TDS on flow as was done to develop the TDS data for the Sweetwater inflow, a multiple regression relationship of TDS on the mean annual flow and month was developed. However, much of the effect of the annual runoff was due to the runoff from the prior year. This result indicates that the water that was in storage in the reservoir was a controlling factor in the TDS of the release. Consequently, the previous year's annual runoff was entered into the multiple regression.

The input TDS for the North Platte and Sweetwater rivers are shown on Figure 30. There were monthly TDS observations for the North Platte throughout the year. There were monthly observations only for the Sweetwater River during water year 1977; as a result, there were no observations in October, which is in water year 1978. Because the predicted values for the North Platte are based on the month-number (Jan. = 1…Dec. =12), the resulting predicted TDS remains constant throughout the month. The observed TDS for the North Platte similarly shows little variation. The predicted TDS for the Sweetwater River is based on flow and varies daily. In both cases the observed and predicted TDS show reasonable agreement with each other. Although the TDS affects the density of water, at the TDS of the inflows to Pathfinder Reservoir, the effect is minimal.

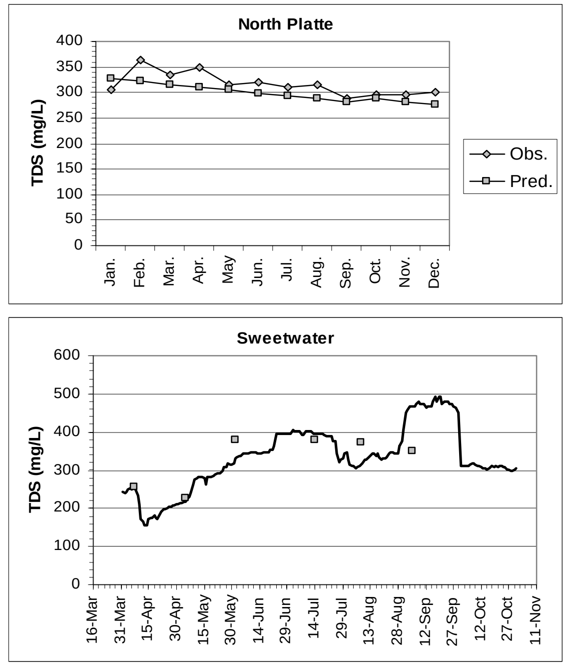


Figure 30. Inflow TDS for the Pathfinder model

Suspended solids can affect the distribution of heat within a reservoir. For this reason, total suspended solids (TSS) are included in CE-THERM-R1. There are no TSS data on the North Platte Rivers during 1977, but there are several observations on the Sweetwater River. There are many more observations on the Sweetwater over the total time that its water quality has been monitored, which include the period of 1964 through the present, although there are some significant gaps in the record. Alternatively, there are only 12 TSS observations from the North Platte upstream from Pathfinder.

Figure 31 shows a scattergram of the TSS data from the Sweetwater River plotted against flow. The usual relationship between TSS and flow, unlike that for TDS, is usually positive. The positive relationship may reflect either the ability of water to transport more sediment as flows increase or an increase in loadings from storms. The scatter in the TSS-flow data on Figure 31 would indicate that whatever is controlling the underlying relationship tends to be inconsistent. Because of the large amount of scatter in data, the r^2 of the regression is relatively low. In the absence of anything better, the relationship shown on Figure 31 was used to develop the TSS of the Sweetwater River inflow to Pathfinder Reservoir.

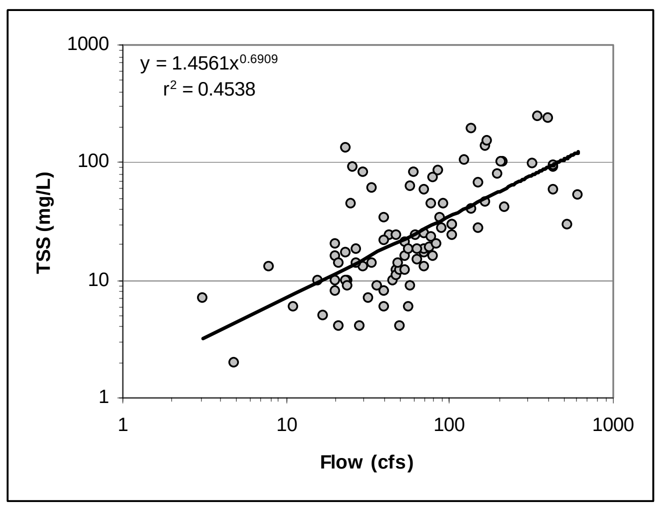


Figure 31. TSS in relation to flow of the Sweetwater River near Alcova

As was the case with temperature and TDS data of the North Platte inflow to Pathfinder Reservoir, the presence of Seminoe Reservoir complicates the TSS regime. However, in the case of TSS, there is a relationship to flow; although the relationship is inverse, rather than positive (Figure 32). An inverse relationship indicates that it is dominated by dilution, which is a characteristic of dissolved solids rather than suspended solids. The inverse relationship between flow and any suspended solids usually indicates that the solids are fine and transport mechanisms are not a factor controlling their concentration. The upstream reservoir acts as a settling basin and removes all but the finest sediments. Figure 32 indicates that TSS concentrations have a relatively small range in the North Platte River upstream from Pathfinder Reservoir (downstream from Seminoe Reservoir).

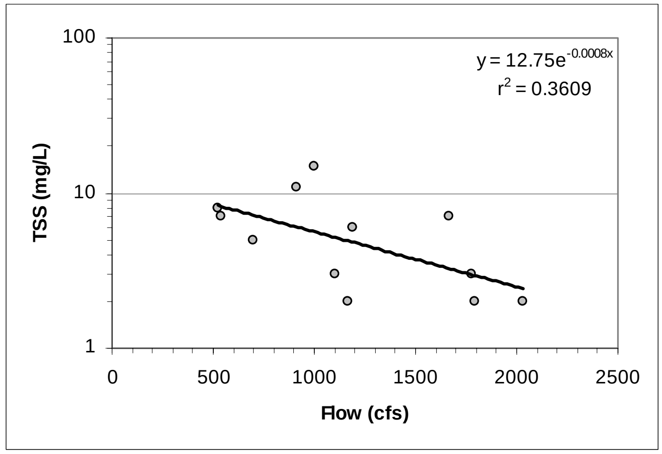


Figure 32. TSS and flow of the North Platte River usptream from Pathfinder Reservoir

As was noted above, the TSS data for the North Platte River at the site upstream from Pathfinder Reservoir are limited (Figure 22). In addition, despite a wide range in the flows, the range in TSS concentrations in the North Platte River at the site is small (compare with Figure 31), but there is still scatter in the data. All of this leads to a comparatively low r^2. Once again in the absence of anything better, the relationship shown on Figure 32 was used to develop the inflow TSS for the North Platte River. Figure 33 shows the TSS concentrations in the modeled tributaries to Pathfinder Reservoir. The TSS of the North Platte River shows the beauty of a regression with a low r^2 – it is unresponsive to changes in the independent variable. In this case that seems to match reality as well, *i.e.* TSS concentrations in the North Platte inflow are never going to be very high. In both the observed data (Figure 32), albeit limited, and the predicted data on figure 33, the maximum TSS is only slightly greater than 10 mg/L.

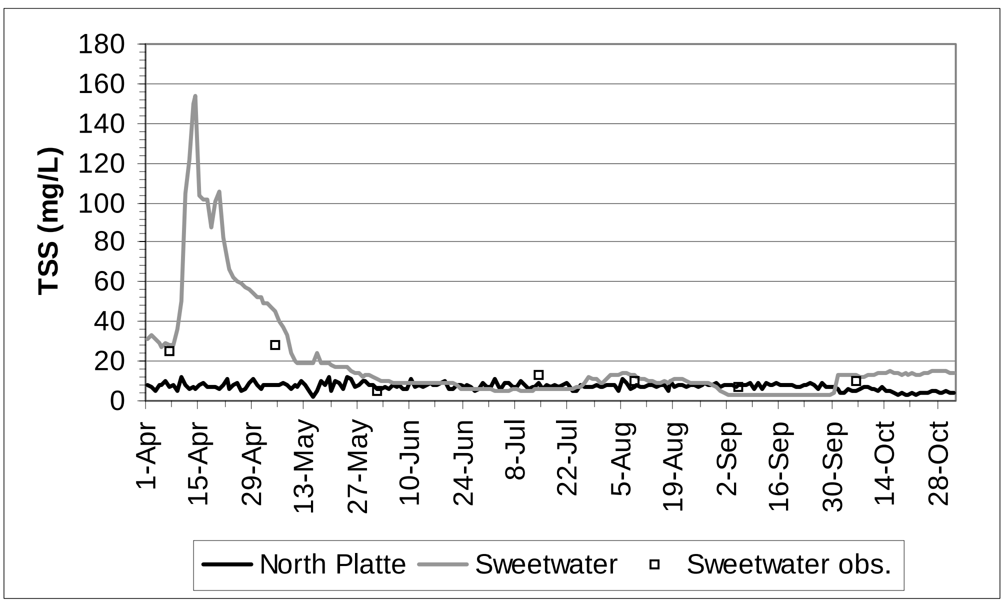


Figure 33. Estimated inflow TSS concentrations during 1977 for the Pathfinder model inflows

There is only 1 large peak in TSS in the Sweetwater River. There are TSS samples at both ends of the peak in the plot. These are not far from the predicted TSS concentrations. What is interesting is the timing of the peak – it occurs in April. This means the peak flow also occurred in April. The peak flow (runoff) was, therefore, both early and brief, the characteristics of a drought year. For the remainder of the period shown on Figure 23, the observed and predicted TSS concentrations in the Sweetwater River are low, and little different from those in the North Platte River.

Mathematical representation of the reservoir

Pathfinder Reservoir is morphometrically rather complex. The surface layout of the reservoir is shown on Figure 34. Pathfinder Dam is located at the head of Fremont Canyon, which is vertical-walled and incised in granite. Downstream from the dam, the canyon extends to the east, before turning to the north to enter Alcova Reservoir. Most of the discharge from the reservoir enters a tunnel that extends on the north side of Fremont Canyon to Fremont Canyon Powerplant, located in Alcova Reservoir below the mouth of the canyon. A bypass release from Fremont Tunnel to Fremont Canyon was established in 2002. The temperatures of the release to the canyon will be evaluated later in the results section of this report.

There are several large embayments in Pathfinder Reservoir. The first is the large embayment to the south of the dam that forms the main body of the reservoir (Figure 34). The Sweetwater River enters the reservoir from the north and forms another rather large embayment in that direction. The North Platte River enters the reservoir from the south. There is also a rather large embayment near where the river enters the reservoir; the southern embayment is the North Platte Arm of the reservoir.

Data and relationships are required to represent the reservoir in the CE-THERM-R1 model. The model requires a spillway even though it is not used in the simulation, as in this case, where only the main outlet is used. The physical dimensions of the outlet are required. The model assumes that the outlets are rectangular. Required data include elevation, height, and width. The angle of the outlet to the main direction of flow is also entered. All of these relate to defining the withdrawal zone of the outlet and therefore are important in defining the outlet temperature.

The volumes of the reservoir at different elevations are not computed directly. The volumes are computed from incremental changes in area. To define the areas, an elevation area curve is entered either in terms of a power curve or a quadratic regression. The elevation-area relationship gave a very poor fit to a power curve. The fit to a quadratic regression was relatively good (Figure 35), although a much better fit could be obtained to a cubic polynomial regression ($R^2 = 0.9999$). The one problem with the fit of the quadratic regression is that in the vicinity of the elevation of the minimum pool (approximately 15 meters), the area of the reservoir is underestimated. To enter a cubic regression, the code would have to be modified. Although the program was recompiled, no changes to the code were made.

The model also requires a relationship between elevation and width. WES (1995) recommends a cross-section 300 meters or more upstream from the dam. Pathfinder Dam is located at the head

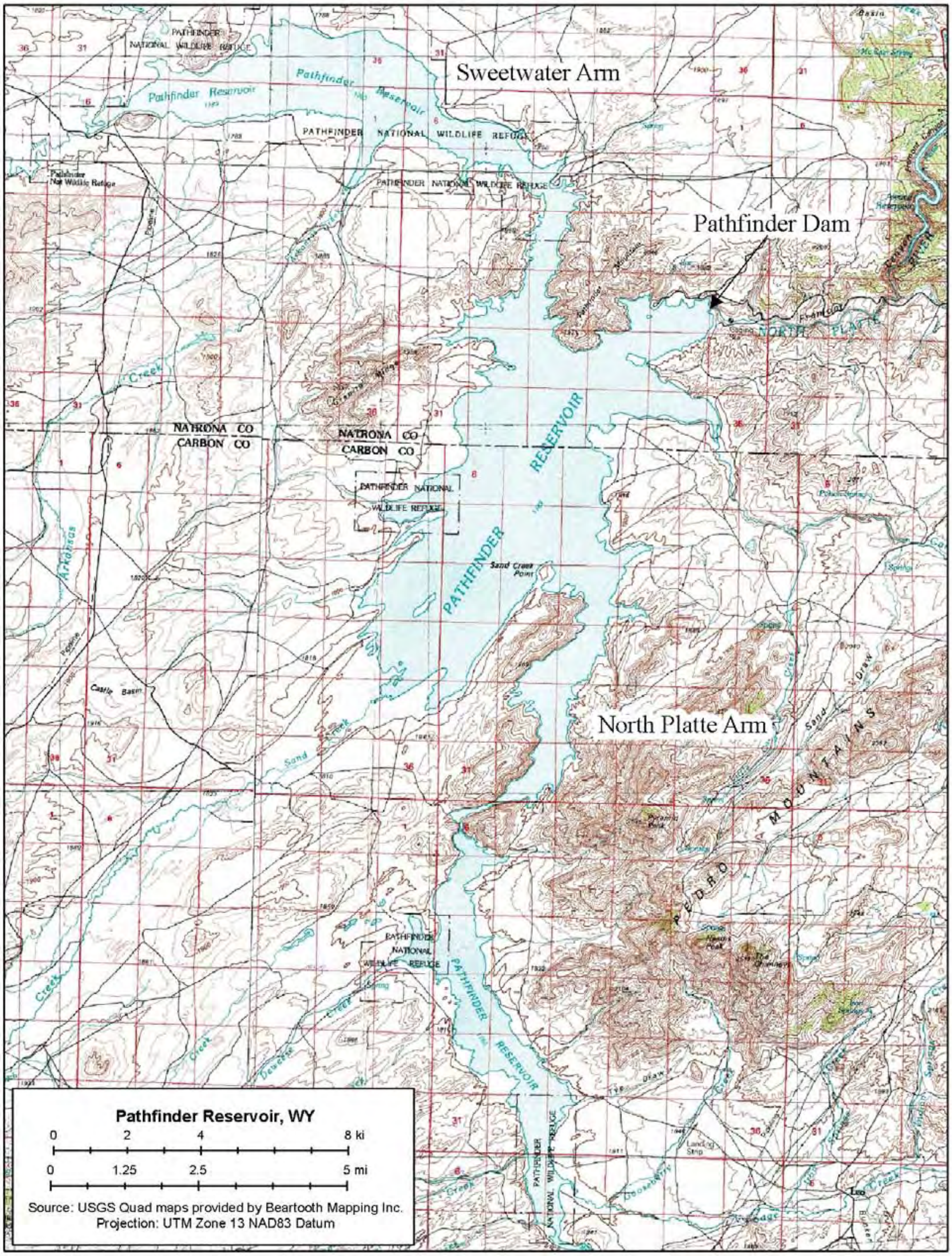

Sweetwater Arm
Pathfinder Dam
North Platte Arm
PATHFINDER RESERVOIR
NATRONA CO
CARBON CO
PATHFINDER NATIONAL WILDLIFE REFUGE
Pathfinder Reservoir, WY
0 2 4 8 ki
0 1.25 2.5 5 mi
Source: USGS Quad maps provided by Beartooth Mapping Inc.
Projection: UTM Zone 13 NAD83 Datum

Figure 34

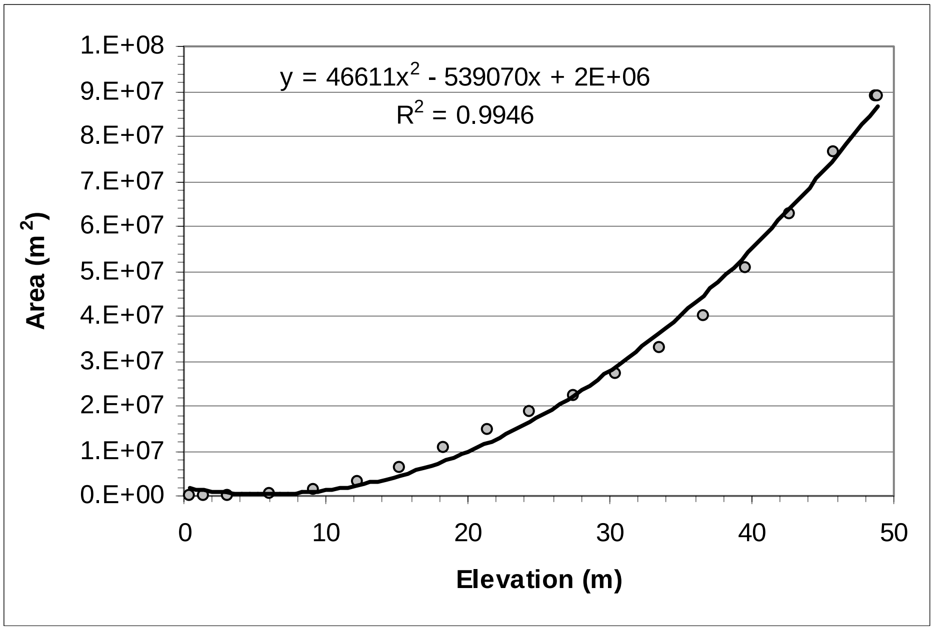


Figure 35: Elevation area curve for Pathfinder Reservoir

of a narrow canyon, and the reservoir opens up considerably not far upstream where the Sweetwater River enters the reservoir. WES (1995) also indicates that a good source of cross-sectional measurements of reservoirs are sediment accumulation studies. The most recent sediment study of Pathfinder Reservoir was conducted in 1953. A copy of the report was obtained from the Sedimentation Section files in the TSC. The nearest cross-section that is shown in detail in the survey report of Seavy and Illk (1953) is near range number 6, which is located 4.3 miles upstream from the dam in a relatively wide area of the reservoir just upstream from the Sweetwater Arm.

Only one option is available to define the elevation-width relationship in the cross-section, a power curve. The relationship developed for the Pathfinder range 6 cross-section is shown on Figure 36. The top width of the cross-section was measured at 8,400 feet (2,661 meters). The lowest elevation in the measured cross-section was at elevation 5,720 feet or about 30 feet above the bottom elevation of the reservoir at the dam. However, the lowest elevation at which an unobstructed width could be measured from the cross-section was at elevation 5,740 feet. This represents the base of the curve shown on Figure 36.

There are a few other characteristics that are required by the model. The reservoir length at average pool content pool was estimated at 12,872 meters (7.7 miles). The latitude and longitude are also needed to estimate the clear sky incoming solar radiation. The radiation is adjusted by the day time average sky cover described above, an atmospheric turbidity parameter, and a shading factor. These latter 2 parameters can be adjusted during calibration to reduce the solar heating at the reservoir surface and consequently the total solar energy available to the reservoir.

An initial reservoir water quality profile is also required by the model. WES (1995) recommends that the initial conditions be defined when the reservoir is isothermal. There are no data on when the reservoir was isothermal in 1977. The earliest profile was measured in June, after

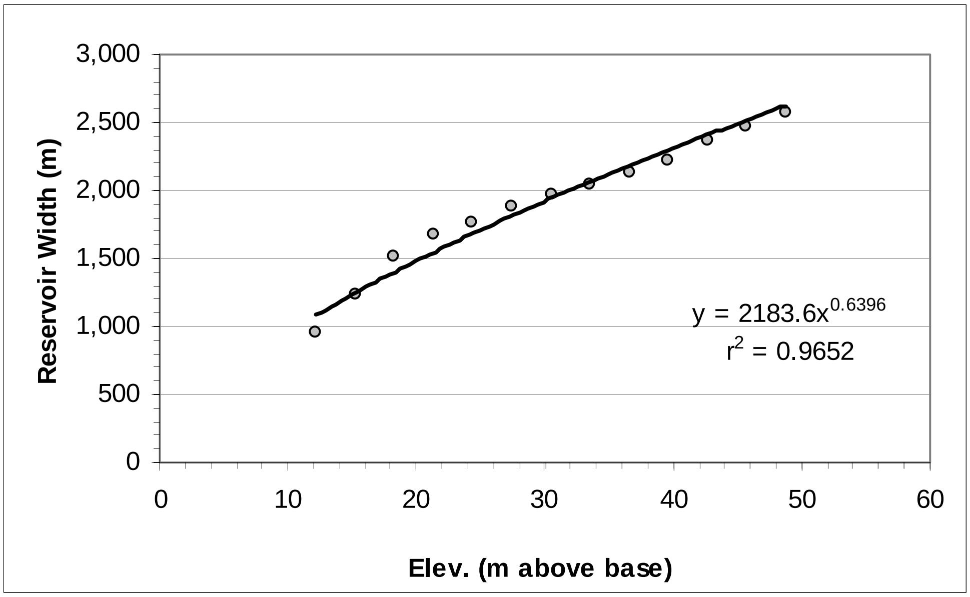


Figure 36. Elevation-width curve for a cross-section located approximately 0.2 miles below (downstream from) sediment range 6 of the 1953 sediment survey

stratification was well established. Based on the assumption that conditions would be similar to Seminoe Reservoir, the temperature data for the Miracle Mile were reviewed. The maximum density of water occurs at a temperature of 4°C (39°F). Isothermal conditions usually prevail at the maximum density of water. The temperature was near 4°C around April 1, 1977, downstream from Seminoe Reservoir. April 1 was used to define the initial conditions in Pathfinder Reservoir. The temperature was set to 4°C in the initial conditions. The TDS was set to 324 mg/L and the TSS was set to 5 mg/L. These are the long-term flow weighted average TDS and TSS of the inflows to Pathfinder Reservoir.

One other datum required by the model is a light extinction coefficient. Sartoris *et al.* (1981) measured light extinction in North Platte reservoirs in 1977 on 5 occasions, including both of the times that surveys were conducted in Pathfinder Reservoir. The light extinction coefficients ranged from 0.46/m to 1.12/m, which range encompassed all of the other measurements. The geometric mean of the 5 values is 0.71/m, which was initially used in the model. This was reduced slightly during calibration to a value of 0.65/m.

Calibration

The first thing that was checked on the first calibration simulation was the elevation-area-capacity data. As was noted above, a quadratic polynomial was used to fit the elevation-area relationship. However, no elevation-capacity relationship was entered. The model used the elevation-area curve and the change in elevation to compute the reservoir volume. Figure 37 shows a comparison of the computed reservoir volumes in meters and cubic meters from the model and the data from the Reclamation records, converted from feet of elevation and acre-feet of storage capacity. The fit is reasonably good. The capacity is underestimated somewhat in the

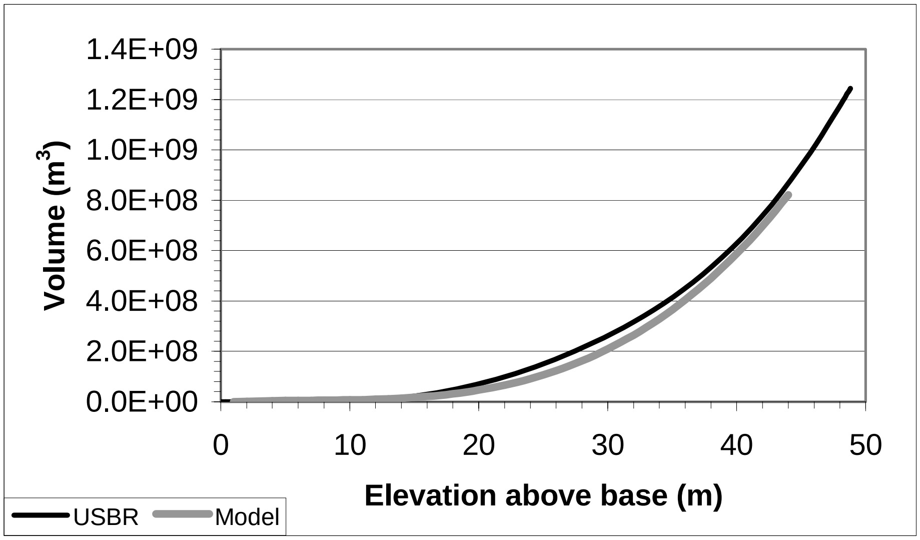


Figure 37. Comparison of Reclamation area-capacity relationship and that computed in the Pathfinder temperature model

section of the curve at 30± meters elevation. The difference was not considered significant enough to try to correct for it. The difference at the upper end of the curves reflects the starting elevation of the 1977 simulation. The reservoir was not full, which is the upper end of the Reclamation curve.

The first several attempts at running the model failed, as is usually the case, because of errors in the input file. The most common error in the file was a disagreement between the coded number of observations of given data types and the actual count of the number of observations presented. Once all of these problems were worked out, the calibration process began. The initial calibration focused on modifying the mixing parameters to get the distribution of heat in the simulated reservoir in reasonable agreement with the observed values. After numerous simulations and adjustments to the mixing coefficients and various parameters that adjust the amount of heat striking the reservoir surface, a reasonable fit to the data was obtained. However, the profiles were still somewhat too warm. The simulations to this point had been run using a set of coefficients for the relationship between wind and evaporation that were developed for Colorado lakes. The substitution of a set of coefficients developed for Australian lakes improved the fit between the simulated and observed temperature profiles considerably. Although a few more adjustments were tried, no better fit was obtained. The final calibration profiles are shown on Figure 38.

The poorest fit of any of the profiles is that for June (Figure 38). The simulated stratification is stronger than what was observed at the time and reflects a somewhat higher surface temperature than was observed. The bottom temperature is also somewhat higher than was observed, but the 2 curves intertwine between elevations of 30-40 meters (depth of 5-15 meters). Some of the difference may reflect a residual effect of the initial conditions. As was noted above, the actual timing of isothermal conditions is uncertain, as is the water temperature when isothermal conditions prevailed. Because the critical period to evaluate effects was later in the summer (July and August), the differences in the June calibration was not considered a major problem.

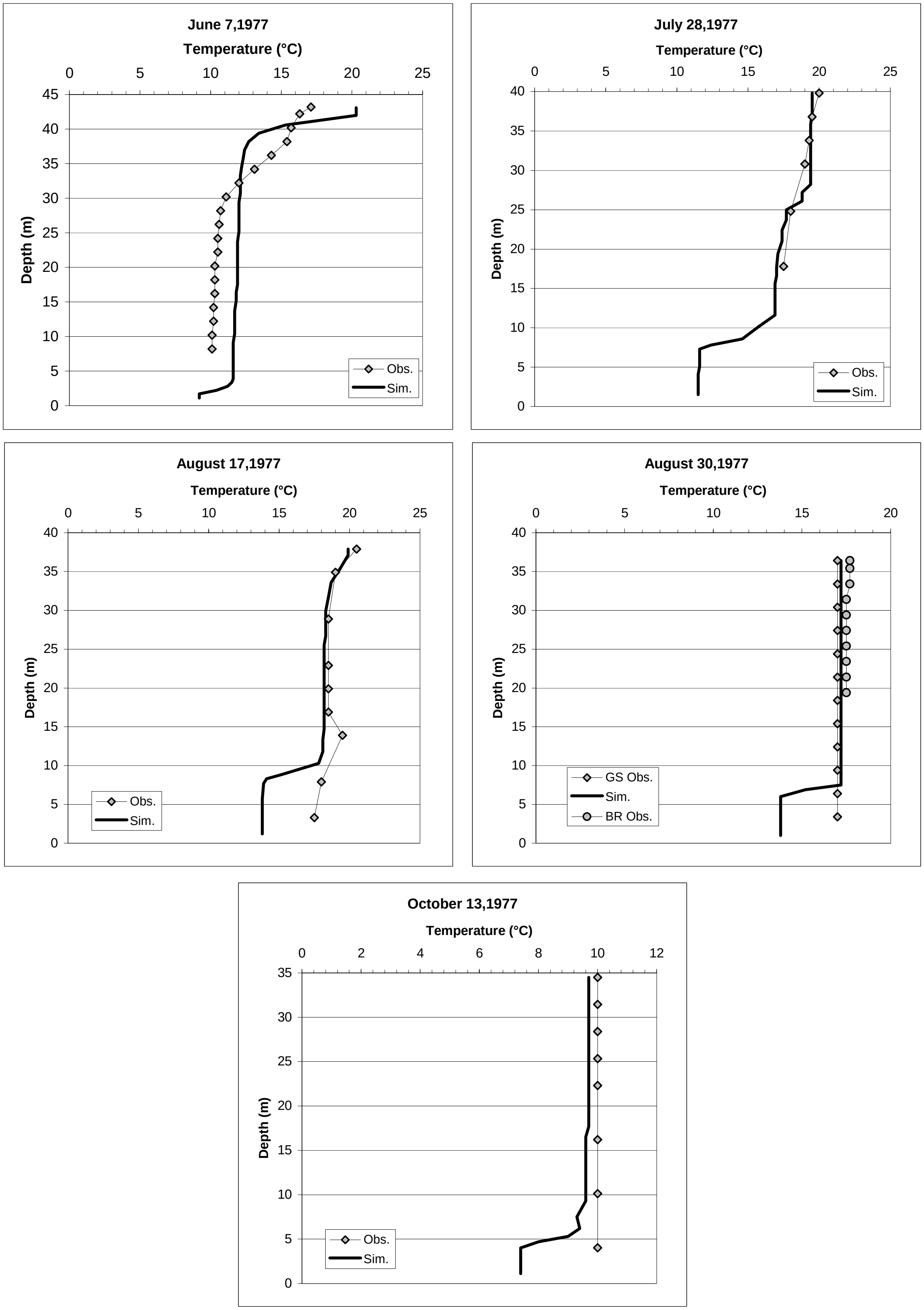


Figure 38: Measured and simulated 1977 temperature profiles for Pathfinder Reservoir

The remaining simulated profiles give an excellent fit to the measured data throughout most of their length. The only departure appears at the lower end of the profile. The simulated profiles appear to be affected by an underflow. The underflow appeared to be due to a density current due to the inflow of cold water and was present from the earliest calibration attempts. The apparent underflow in the simulated, but not in the measured profiles, led to the switch in the way that the North Platte inflow temperature was estimated (see above). However, although the switch from a regression on air temperature to the seasonal regressions on the Sweetwater temperatures did lead to a warmer North Platte inflow, it did not make any difference in the profiles.

Further attempts to remove the apparent density layer by changing the density parameter and other mixing coefficients failed to change the profiles shown on Figure 38. A further review of the output from the model showed that the density layer included the highest concentrations of TSS (Figure 39). Although the TSS concentration in the density layer is not very high, the layer contains virtually all of the suspended solids in the profile. What the profiles appear to be showing is an isolated layer of cold, turbid water below the sill of the intake to the outlet tunnel. Sheppard (1959) concluded on the basis of the low density sediments deposited near the dam that there were density flows that carried fine sediment through the reservoir. The 1959 study was conducted at the time that the reservoir had been evacuated to facilitate construction of the tunnel for the Fremont Canyon Powerplant and the sediments could be accessed directly (*ibid.*). The cold, turbid layer amounts to a relatively small volume of water trapped below the level of the intake to the outlet. Because the density layer in the model is below the level of the outlet, it can only affect the overlying water and the releases by upward diffusion as the underflow pushes the layer upward. A review of the model output indicates that the withdrawal zone does not pull water from that layer, further indicating its isolation from the remainder of the reservoir. Because of the isolated nature of the layers, the calibration shown on Figure 38 was accepted as final.

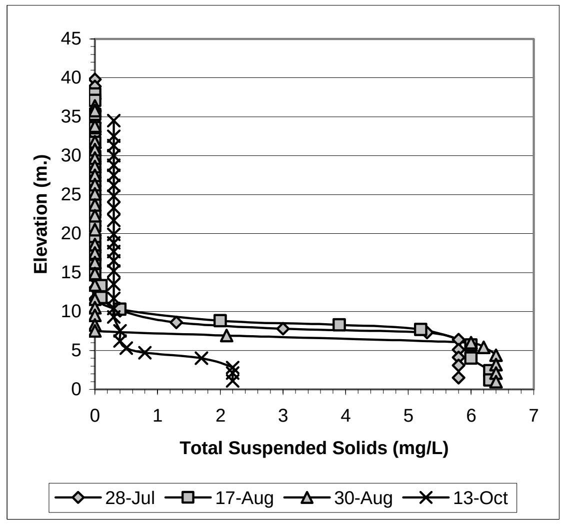


Figure 39. Simulated TSS profiles in Pathfinder Reservoir

The profile for October 13 appears to be rather cooler than the observed data (Figure 38). However, the observed data were USGS published data which were rounded to the nearest whole degree, in this case, 10°C. The simulated profile is at 9.7°C, which rounds to 10°C. The simulation is close enough to the published values to be the same as those published values. This further validates not only the calibration, but the heat budget as well.

Results

The Pathfinder Reservoir model was run based on the output from the North Platte Basin EIS Operations Model. As a comparison with the calibration, the simulated Present Condition flows and water levels for 1977 were entered into the model. Profiles from 2 dates from each output file are shown on Figure 40. Frequently models developed for impact assessment show the greatest difference between observed and the modeled Present Condition. This does not seem to be the case with the Pathfinder Reservoir temperature model nor the underlying North Platte EIS Operations Model. There are no major differences between the 2 sets of profiles.

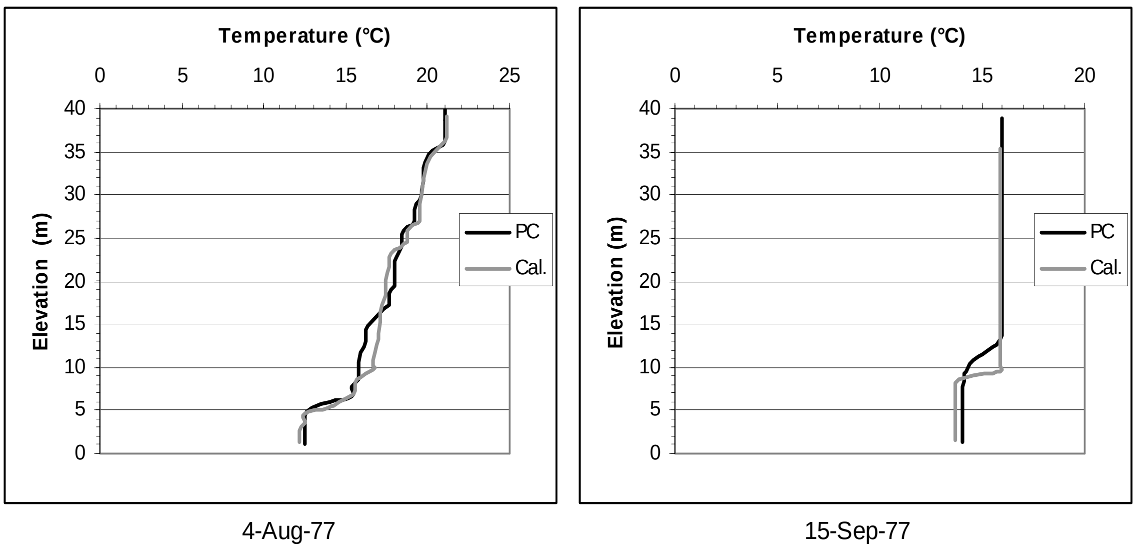


Figure 40. Comparison between simulated temperature profiles for 1977 in Pathfinder Reservoir from the final calibration (Cal.) and the Present Condition (PC)

One of the reasons for undertaking the model study was to evaluate the effects of drawdown on the temperature structure of the reservoir. The Present Condition does not show any drawdown on Figure 40. There is little difference in the elevations in the August profiles, while the elevation for the Present Condition profile is about 3 meters higher than was the case historically. Consequently, the profiles from both dates are intertwined. The one thing of note is that the thermocline, which appears to be a reflection of the influence of TSS that was discussed above in the calibration section, is shallower and weaker than in the calibration run.

In the following sections of this report, the model output from each of the alternatives will be compared to that of the Present Condition. This type of comparison puts everything on the same basis. However, as the preferred alternative is the Governance Committee Alternative, its results will be the focus of the report and be reported in somewhat more detail than the will be the results for the other alternatives.

Effects of the Governance Committee Alternative – Temperature Profiles

The comparison between the model output with the Present Condition results in the form of temperature profiles for selected dates will be based on significant occurrences in the reservoir related to the Governance Committee operations. These same dates will become benchmarks for the comparison of all the alternatives with the Present Condition.

Temperature profiles from the Governance Committee Alternative and Present Condition model output for 1961 are shown on Figure 41. The profiles for the 2 alternatives (the Present Condition represents the No Action Alternative) shown for late July are virtually the same. Although there are small differences in temperature at points in the profile, there is no overall difference in the size of the reservoir layers, *i.e.* epilimnion, metalimnion (thermocline[1]), and

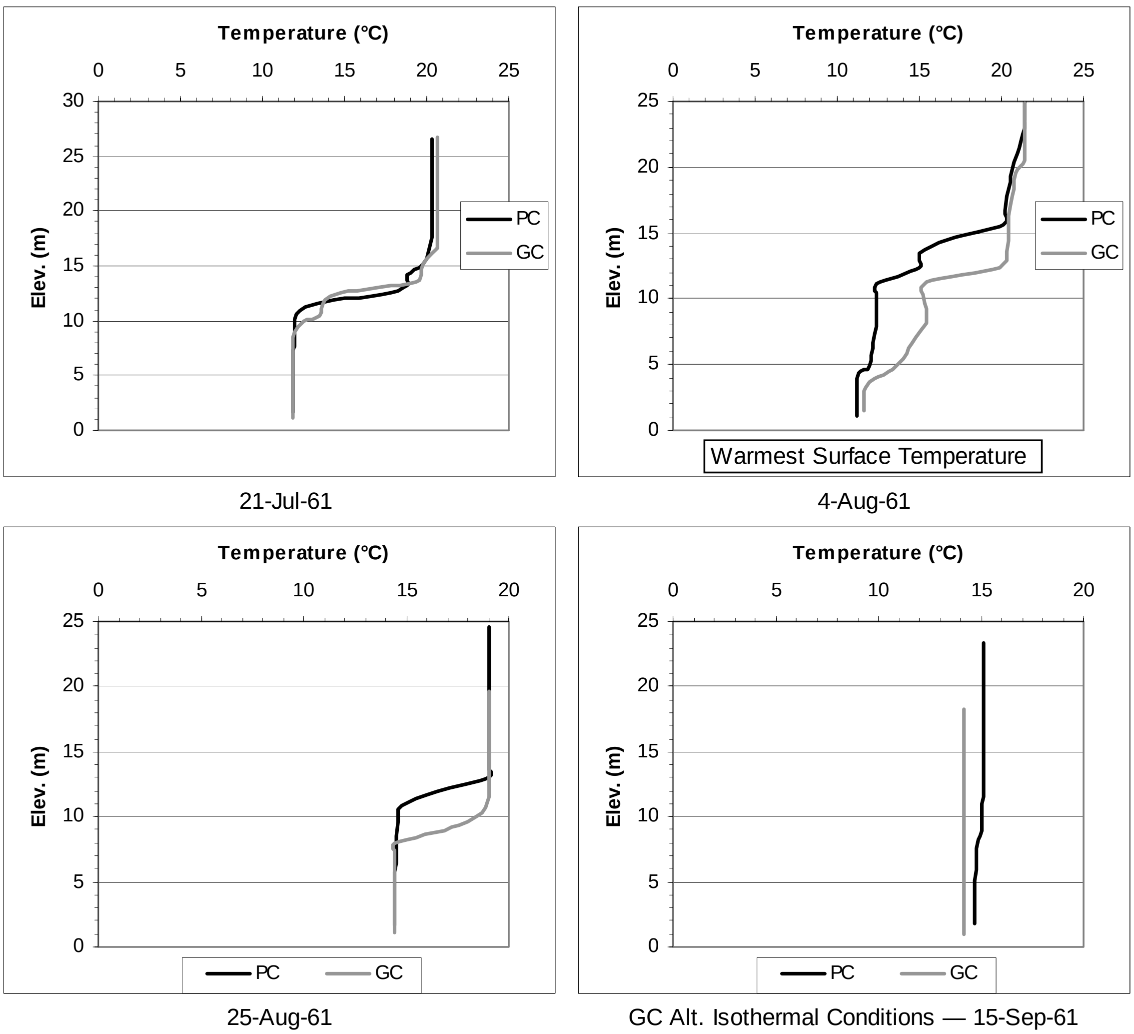


Figure 41. Governance Committee Alternative and Present Condition Temperature Profiles on 4 dates under simulated 1961 conditions

[1] The metalimnion is layer in which there is a rapid temperature change, while the thermocline is the layer within the metalimnion in which the maximum temperature change occurs and is traditionally defined as having a minimum temperature change of 1°C per meter of depth

hypolimnion. At this point in the year, there is no real difference in the sizes of the reservoir pools. Under both conditions, the reservoir is over 25 meters (82 feet) deep.

By early August, differences in the profiles begin to appear. As is noted on Figure 41, August 4 represents the date in the model on which the warmest surface temperature would be observed. However, there is no difference from the surface temperature of the Present Condition profile. Interestingly, the total depth of the reservoir is greater on August 4 with the Governance Committee Alternative than under the Present Condition. This means that the reservoir drawdown is going to be relatively rapid with the Governance Committee Alternative. As was shown in Table 14, the reservoir is not drawn down to its minimum operating level in August under the Present Condition. The main differences in the August 4 profiles between the alternatives on Figure 41 concerns the size of the epilimnion and the depth of the metalimnion. The epilimnion is much larger.

As a result there is much more warm water in the reservoir under the Governance Committee Alternative. The hypolimnion is also much smaller under the Governance Committee Alternative. This distribution of heat in the reservoir is the real issue for coldwater fish. Under these conditions, the hypolimnion is much more likely to become anoxic than would be the case with a larger layer. At the onset of stratification, the hypolimnion becomes isolated from atmospheric oxygen exchange. So whatever oxygen is present at the onset of stratification is all that will be there until stratification breaks down. Due primarily to bacterial respiration in the sediments, oxygen in the hypolimnion becomes depleted. If respiration is sufficiently great, the entire hypolimnion can become anoxic. If the hypolimnion is the only area of the reservoir that has temperatures suitable for coldwater fish, those fish will have to either leave the reservoir or die. In either case, the coldwater fishery in the reservoir would be eliminated.

Another consideration in the reservoir and temperature model compatibility concerns the time scale of the 2 models. The reservoir operations are taken from the North Platte EIS Operations Model, which operates on a monthly time step. The temperature model operates on a daily time step. The reservoir inflows from the North Platte River and its releases are taken as monthly data from that model. The daily flows are represented by the mean monthly flow from the operations model and remain constant throughout the month. Consequently, any drawdown occurs at a constant rate over the course of a month.

By late August, the draw down of the reservoir becomes more apparent. The Present Condition Profile still has an elevation (equivalent to the total reservoir depth) of about 25 meters. However, under the Governance Committee Alternative, the total reservoir depth is just under 20 meters (66 feet). Nevertheless, the temperature profiles are similar in form. The only difference is the depth of the metalimnion. Under the Governance Committee Alternative, the metalimnion is about 3 meters (10 feet) deeper, once again making for a smaller hypolimnion. The smaller hypolimnion may not be a major factor, in that the entire profile is within what is usually considered suitable for coldwater fish. The epilimnetic temperature, which is the highest in the reservoir, is 19°C (66°F).

Under both of the alternatives, the epilimnetic and hypolimnetic temperatures are the same. The difference is the increase in the warmer water that constitutes the epilimnion. As a consequence, there is more heat in the reservoir under the Governance Committee Alternative, which is what was anticipated before undertaking the study. What was not anticipated was how the additional heat would be manifested in the temperature structure of the reservoir. Instead of the temperatures being higher in the smaller reservoir pool, the distribution of heat is different. Actually, because the reservoir is so much smaller, there may be less total heat in the reservoir under the Governance Committee Alternative.

The last set of profiles (September 15) represents the point at which the reservoir became isothermal under the Governance Committee Alternative. The difference in the total depth of the reservoir is still apparent in the profiles. As was shown in Table 14, the reservoir was drawn down to its minimum operating level in September in one year. However, that year was not under the conditions represented by 1961. Under the Governance Committee Alternative the reservoir is isothermal at a temperature of about 14°C (57°F), while under the present condition the temperature is about 1°C (1.8°F) warmer. The Present Condition profile is not strictly isothermal; there is a difference of a few tenths of a degree C between the surface and the bottom of the profile.

The difference in the temperatures in the 2 profiles indicates another difference in the heat budget of the reservoir between the 2 operations. Although there was more heat in the reservoir in August, there is less in September. What this indicates is that the smaller pool of the Governance Committee Alternative loses heat more rapidly than the larger pool of the Present Condition.

Figure 42 shows the results of the 1964 temperature simulation for the Governance Committee Alternative in comparison with the Present Condition. The first set of profiles is for July 21. The drawdown associated with the Governance Committee Alternative operations is already apparent. The reservoir is about 5 meters (16 feet) lower under the Governance Committee Alternative operation. Although the thermocline appears deeper under the Governance Committee Alternative operation, it is not. In the Governance Committee Alternative profile, the epilimnion extends to a depth of about 10 meters (30 feet), while in the Present Condition profile, it extends to a depth of about 13 meters (43 feet). On the whole, the epilimnion and hypolimnion are smaller and stratification is stronger in the Governance Committee Alternative profile in comparison with the Present Condition profile. The strength of stratification is illustrated by the maximum temperature change in the thermocline. In the Governance Committee Alternative profile, the maximum temperature change is nearly 4°C (7°F) in less than 1 meter, while the maximum temperature change in the Present Condition profile is 3°C (5°F) in a little over 1 meter. On the other hand, there is no difference in the epilimnetic temperatures (20.3°C or 68.5°F). The hypolimnion is slightly warmer in the Governance Committee Alternative profile (Figure 42).

Although July 28 is only 1 week later than the date of the preceding profile, the profile for that date is included because stratification in the Governance Committee Alternative profile became even stronger in the intervening week and was the strongest observed in its 1964 set of temperature profiles (Figure 42). The maximum temperature change in the July 28 profile for the Governance Committee Alternative is 4.5°C (8.1°F) in less than 1 meter of depth, while in the profile for the Present Condition, the maximum change is 2.1°C (3.8°F), also in less than 1

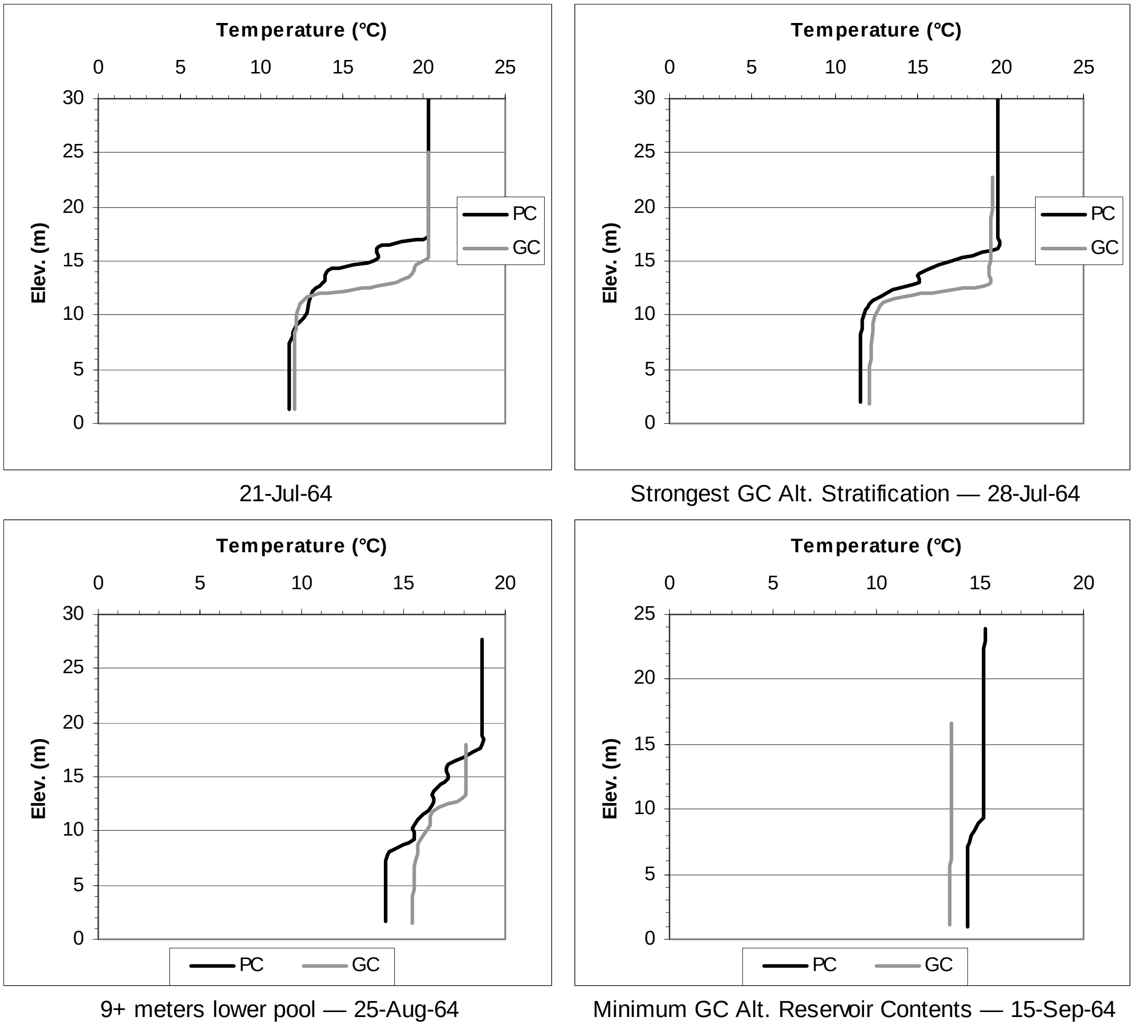


Figure 42. Governance Committee Alternative and Present Condition temperature profiles on 4 dates under simulated 1964 conditions

meter of depth. The strength of the thermocline affects heat and gas (*e.g.* O_2) exchange between the epilimnion and the hypolimnion. Although it was noted above that the hypolimnion becomes isolated once stratification becomes established, there would still be some exchange due to diffusion. As stratification becomes stronger, diffusion decreases, and the degree of isolation increases.

There is a diffusion coefficient in CE-THERM-R1. It's function is somewhat similar to the above, but it also differs in some respects. Profile plots of the diffusion coefficients on July 28, 1964, for the Present Condition and the Governance Committee Alternative are shown on the left hand side of Figure 43. As can be seen on Figure 43, the diffusion coefficients for the Present Condition are actually smaller than those of the Governance Committee Alternative. However, the diffusion coefficients in CE-THERM-R1 do not restrict movement of heat or other constituents between layers, They are used in distributing inflows and outflows, as is also shown on Figure 43. The small diffusion coefficients for the Present Condition affect the distribution of outflows, but do not greatly affect the outflows, mainly because the very small diffusion

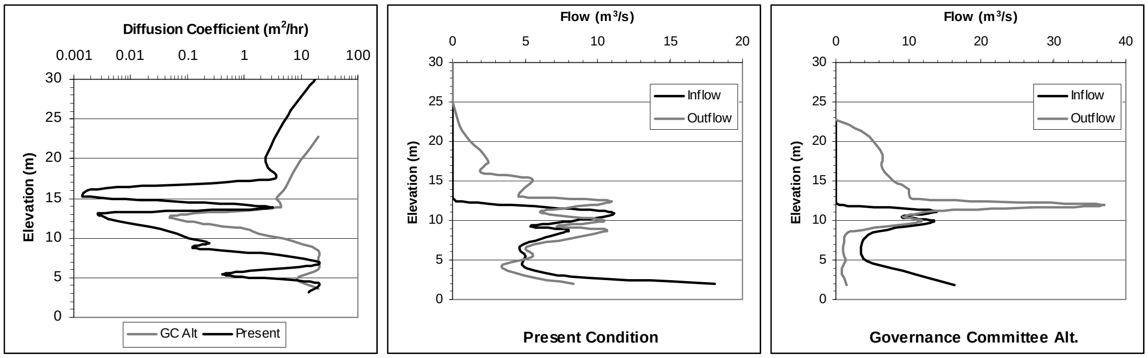


Figure 43. Profiles of layer diffusion coefficients and inflows and outflows for the Present Condition and the Governance Committee Alternative as simulated on July 28, 1964

coefficients occur in more than 1 layer. The temperature of the inflows is a factor in distributing the inflows, as is the TDS and TSS. All 3 are factored into estimating densities. Nevertheless, there is a large underflow in the Present Condition inflow-outflow profiles. The temperature of the North Platte inflow at the time was 11.2°C (52°C), which was cooler than any point in the reservoir profile. This difference appears to be mainly responsible for the decrease in temperature observed in the bottom of the Present Condition profile that was observed between July 21 and 28. Alternatively, there was no cooling in the bottom of the temperature profile between dates for the Governance Committee Alternative. The inflow is distributed in a similar manner, except that the underflow is smaller, probably reflecting the fact that the inflow for the Governance Committee Alternative is also much smaller.

Figure 43 shows that outflows are withdrawn from a wide range of layers in the reservoir profile for the Present Condition. Nevertheless, the majority of the outflow is still withdrawn from below the thermocline (refer to Figure 42). However, the withdrawal from the reservoir under the Governance Committee Alternative is almost all from a layer located just below the thermocline, although there is some withdrawal from all layers in the reservoir. The overwhelmingly large withdrawal from the lower part of the thermocline would serve to erode (weaken) it.

Although the reservoir has not been drawn completely down to its minimum operating level by August 25 under the Governance Committee Alternative operation, it is getting close and would reach that level by the end of the month. The total depth of the reservoir under the Governance Committee Alternative is more than 9 meters below that of the Present Condition (Figure 42). Although the reservoir is drawn down to its minimum operating level under the Present Condition operation in 1964, that level is not reached until the end of September. The main differences in the profiles under the 2 operations include a warmer hypolimnion and a cooler epilimnion in the Governance Committee Alternative profile. This smaller difference in temperature between the surface and bottom layers should mean that the reservoir will mix earlier under the Governance Committee Alternative. However, there is evidence of instability in the profile of the Present Condition as well.

The last set of 1964 profiles on Figure 42 is based on the output for September 15. Under the Governance Committee Alternative, the reservoir is isothermal by that date. Although profiles were only output from the model on a weekly basis, the outlet temperatures, which were output daily, indicate that complete mixing probably occurred on September 15. The Present Condition profile still shows some difference between the surface and bottom of the temperature profile. However, the main difference in the profiles is that the Governance Committee Alternative profile is uniformly cooler that that of the Present Condition. This difference is a further exemplification of the more rapid cooling of the smaller Governance Committee Alternative reservoir.

Effects of the Governance Committee Alternative – Fremont Canyon Bypass

The intake to the outlet of Pathfinder Reservoir is located between elevations 5715 and 5733. Taking the difference from the base of the dam (5690 feet), which is the base of the elevations as expressed in the preceding profile plots, the top of the intake is at an elevation of 43 feet or 13.1 meters. The total depth of the reservoir when storage reaches 31,400 acre-feet would be 56 feet (17.1 meters). At the minimum reservoir content of 31,400 acre-feet of storage (equivalent to elevation 5746 feet), the top of the intake would be submerged to a depth of 13 feet (4 meters) and would be drawing water from the upper layers of the reservoir. As was noted above, the withdrawal zone to the intake extends both above and below the level of the intake; so the discharge would reflect the temperature of the mixed water from the entire withdrawal zone. These are the release temperatures that will be used to compare alternatives.

Figure 44 shows the daily temperature of the Fremont Canon Powerplant bypass from April 1 through the end of October 1961 for the Present Condition and the Governance Committee Alternative operations. Through most of the year, the bypass flow temperatures are the same for either of the operations. There is a brief period in early May and another period in late May through early June that the bypass temperature under the Governance Committee Alternative operations is warmer than that of the Present Condition. There is a much longer period beginning around the first of August and ending in early September when that pattern is repeated. Through most of September, the temperature of the bypass is lower under the Governance Committee Alternative operations.

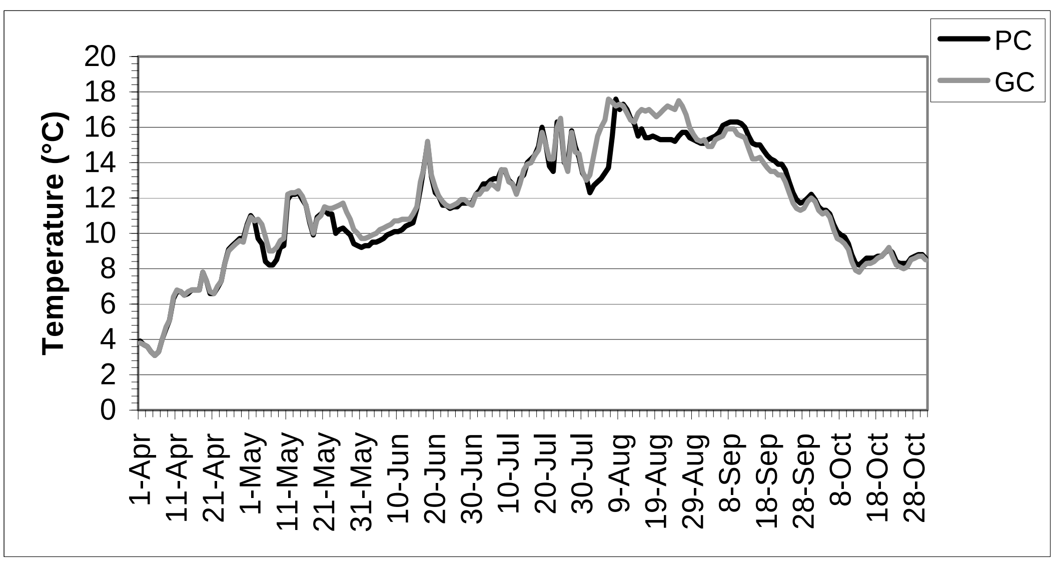


Figure 44. Simulated temperature of Fremont Canyon Powerplant bypass under 1961 conditions and operations of the Present Condition and the Governance Committee Alternative

The most important result shown on Figure 44 is that the maximum temperatures for either the operations of the Present Condition or the Governance Committee Alternative are less than 18°C (64°F). The maximum temperature for each of the operations occurs in early August, but it occurs again in late August under the governance Committee Alternative operation. In either case, the maximum release temperature occurs as a short-lived spike. It appears that the temperature of the Fremont Canyon Powerplant bypass is suitable for coldwater fish throughout the year.

Figure 45 shows a similar plot of the bypass temperatures under the 1964 conditions. Under those conditions, the temperatures are about the same for the 2 operations until early July, at which time the temperature of the Governance Committee Alternative operation becomes noticeably warmer (by about 2°C [3.6°F]) than that of the Present Condition. The temperature with the Governance Committee Alternative operation remains higher until early August. In late August, there is another difference in the bypass temperatures for the 2 operations, but this time the Governance Committee Alternative operation produces a slightly cooler release. In 1964, the maximum temperature of the release is somewhat lower than was the case in the 1961 simulation.

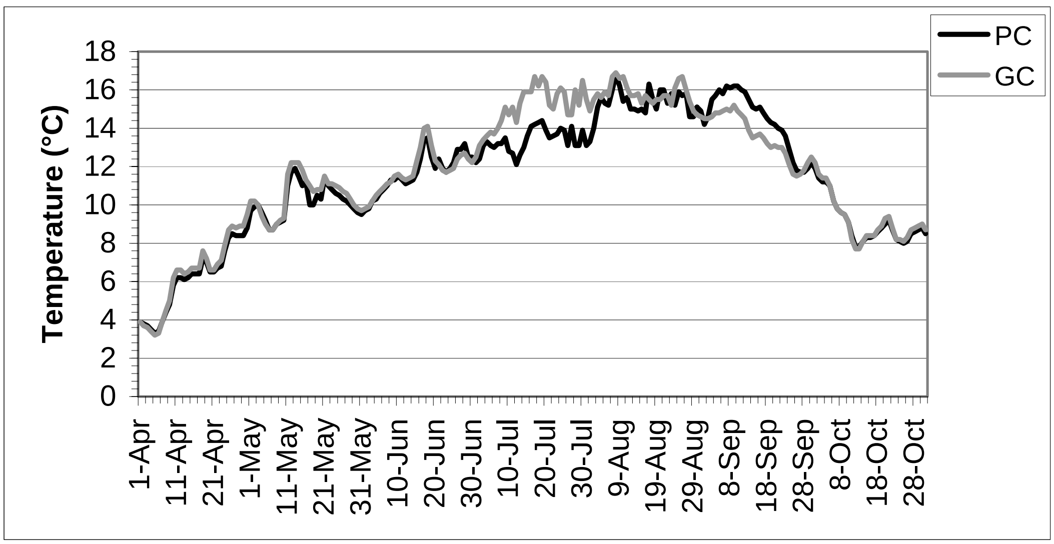


Figure 45. Simulated temperature of Fremont Canyon Powerplant bypass under 1964 conditions under operations of the Present Condition and the Governance Committee Alternative

The peak temperatures of the Pathfinder Dam releases under either of the operations are less than 17°C (63°F). As was the case with the 1961 simulation, the simulated 1964 temperatures of the Fremont Canyon Powerplant bypasses should be suitable for supporting coldwater fish.

Effects of the Water Emphasis Alternative – Temperature Profiles

The 1961 temperature model results for the Water Emphasis Alternative are shown on Figure 46, along with the comparable profiles for the Present Condition. The dates for the plots are the

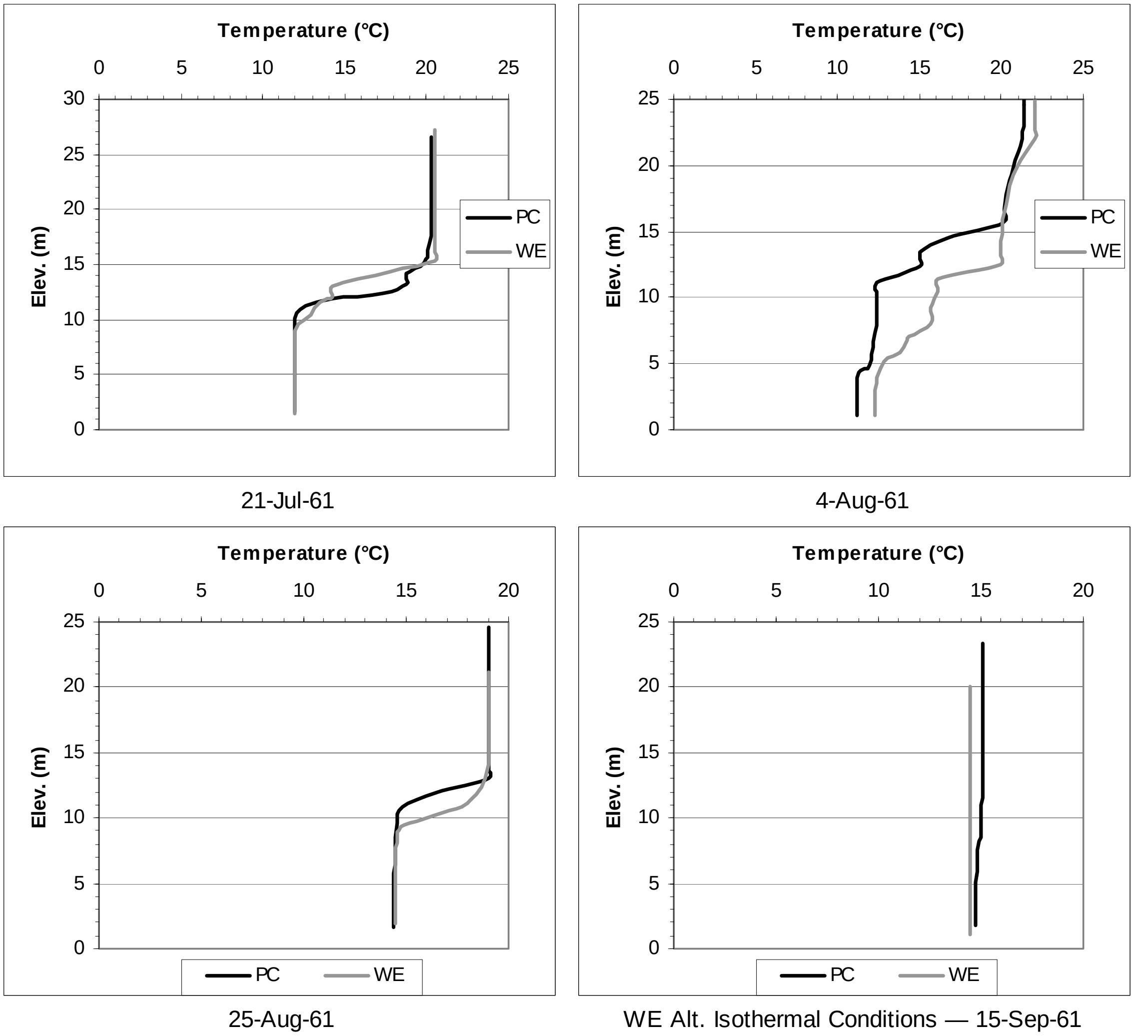


Figure 46. Comparison of selected temperature profiles in Pathfinder Reservoir in simulated 1961 conditions for the Present Condition and the Water Emphasis alternative

same as those used to develop Figure 28. In many respects, the 1961 results for the Water Emphasis Alternative are similar to those of the Governance Committee Alternative.

In late July of the 1961 simulation, the hypolimnion is a bit larger for the Water Emphasis Alternative in comparison with the Present Condition. At the same time, the epilimnion is slightly warmer (by about 0.2°C). The main difference in the 2 profiles is in the metalimnion. The Present Condition profile shows a jog in the temperature distribution in the upper part of the metalimnion, while the Water Emphasis Alternative profile shows the anomaly at the lower end of the profile. The differences in the profiles have to do with the varying influences of the inflows and outflow. In the Present Condition simulation, the inflow is primarily distributed near the lower edge of the metalimnion. The withdrawal zone in the Present Condition simulation extends both above and below the metalimnion with most of the water pulled from the edges of the metalimnion. On the other hand, the inflow in the Water Emphasis Alternative simulation is primarily along the bottom of the reservoir at the base of the profile. The withdrawal zone from the Water Emphasis Alternative is almost entirely from below the metalimnion. Consequently, the lower part of the Water Emphasis Alternative metalimnion is affected more than that of the Present Condition.

Both of the August 4 profiles for 1961 show some deviation from the more normal chair shape of the typical temperature profile of a stratified reservoir. In the case of the Present Condition profile, the primary influence is from the location of the inflow, just below the bottom of the metalimnion. Alternatively, in the case of the Water Emphasis Alternative profile, both the inflow and the outflow are primarily located just below the metalimnion. In effect, there is a current created just below the metalimnion. This current is what is being shown in the lower part of the metalimnion (or the upper part of the hypolimnion). In other words, to a great extent, the inflow is being passed unmixed through the reservoir on August 4 in the Water Emphasis Alternative simulation.

The reason for showing the August 4 profile for the Governance Committee Alternative was that the warmest surface temperature was observed on that date. This is also true of the Water Emphasis Alternative profile, *i.e.* 22°C (72°F). If in fact there is a deep water current in the reservoir on that date for the Water Emphasis Alternative, this would provide a layer of reasonably well oxygenated flow through the upper part of the hypolimnion and help the coldwater fish survive. Although the temperature profile for the Governance Committee Alternative looks somewhat like that of the Water Emphasis Alternative, the distribution of flow is different and the same result may not apply to the Governance Committee Alternative.

By August 25, the reservoir has been drawn down to near its minimum operating level in the Water Emphasis Alternative simulation. The profile assumes a more classic shape. The inflow is entirely into the hypolimnion and the epilimnion has cooled to 19°C (66°F). The epilimnetic and hypolimnetic temperatures in the Present Condition profile are the same as those of the Water Emphasis Alternative profile. There is a difference in the strength of stratification in the 2 profiles, with that of the Present Condition being somewhat stronger on August 25. The main factor appears to be that the reservoir content in the Present Condition is much larger.

By September 15, the reservoir is isothermal under the Water Emphasis Alternative operation. It is nearly so under the Present Condition operation as well, but the temperature difference is only 0.4°C (0.7°F). The temperature under the Water Emphasis Alternative operation is slightly cooler than the lowest temperature in the Present Condition profile. This is a further example of the more rapid cooling of the smaller reservoir pool.

Figure 47 shows temperature profiles for the 1964 simulations of the Water Emphasis Alternative and the Present Condition. The profiles on July 21 and 28 show little difference. Unlike the Governance Committee Alternative simulation, which showed the strongest stratification on July 28, the strongest stratification in the Water Emphasis Alternative simulation occurred on July 21. The similarities between the Present Condition and the Water Emphasis Alternative profiles appears to reflect the similarity in the total depth of the reservoir under their respective operations. The maximum epilimnetic temperatures in the 2 simulations are also about the same at 20°C (68°F).

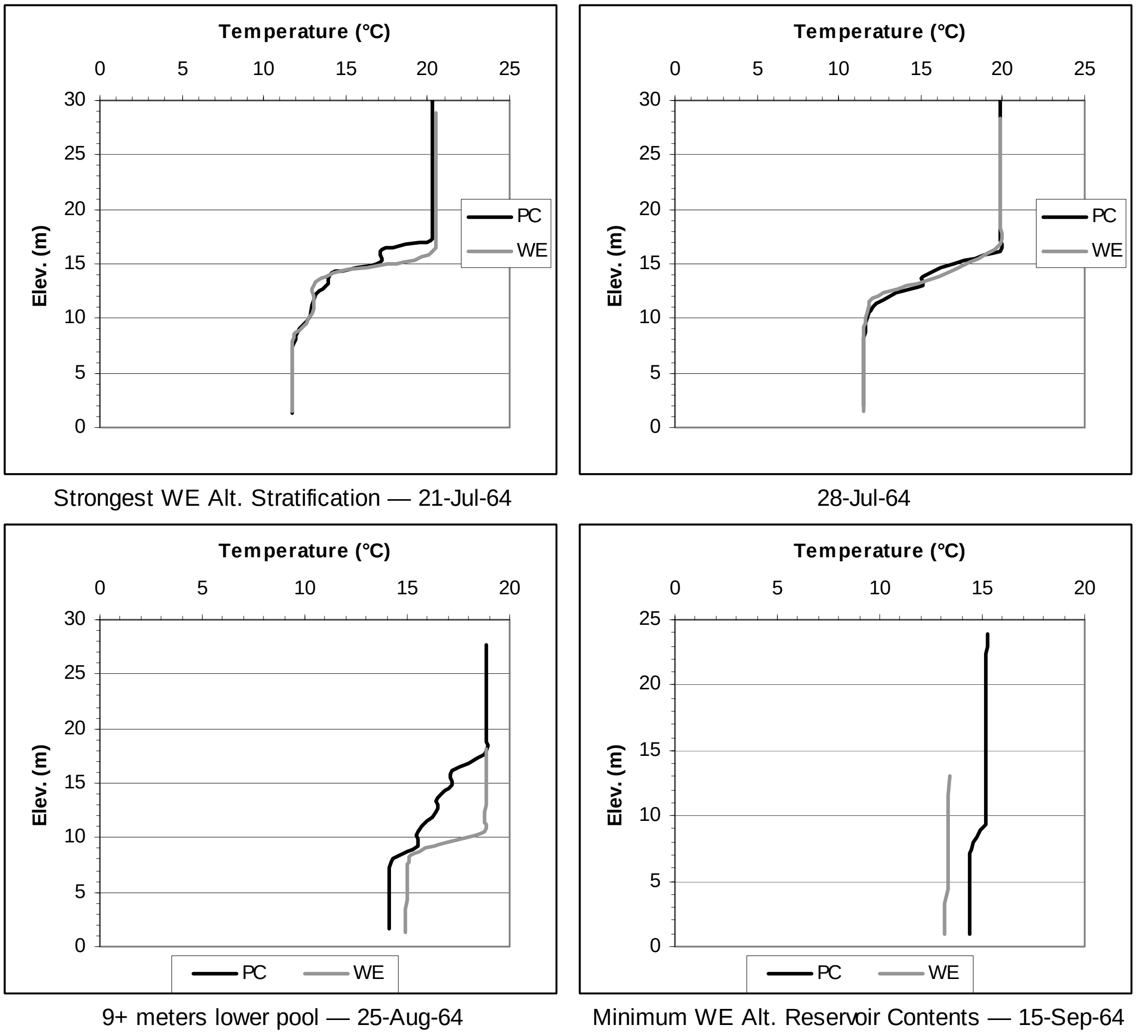


Figure 47. Comparison of selected temperature profiles in Pathfinder Reservoir in simulated 1964 conditions for the Present Condition and the Water Emphasis alternative

By August 25 in the 1964 simulations, there is a large difference in the total depths of the reservoir, with the water surface in the Present Condition over 9 meters (30 feet) greater than that of the Water Emphasis Alternative (Figure 47). Stratification is much stronger in the Water Emphasis Alternative profile, and the hypolimnion is much smaller. However, the temperature is low enough throughout either profile to support coldwater fish. In both cases the maximum temperature in the profile is at the surface and is 18.9°C (66°F).

On September 15, the reservoir is at its minimum operating pool in the Water Emphasis Alternative Water Emphasis Alternative operation. The temperature is nearly isothermal, with only a 0.1°C difference below the surface. In the Present Condition profile, there is still some weak stratification around 8-9 meters elevation, *i.e.* 0.6°C/m. Once again, the Water Emphasis Alternative temperature profile is noticeably cooler than that of the Present Condition.

Effects of the Water Emphasis Alternative – Fremont Canyon Bypass

Figure 48 shows a time series plot of the outlet temperature at the Fremont Canyon bypass for the Present Condition and the Water Emphasis Alternative. For most of the year (April 1 through October 31), there is little difference in the 2 sets of temperature data. The only evident difference occurs during August. The peak temperatures are about the same and both occur in mid-August. Immediately prior to and following the peak temperatures, the outlet temperature is somewhat warmer in the Water Emphasis Alternative operation. In both cases the maximum temperature is 18°C (64°F), which should be suitable for supporting coldwater fish downstream in the Cardwell fishery.

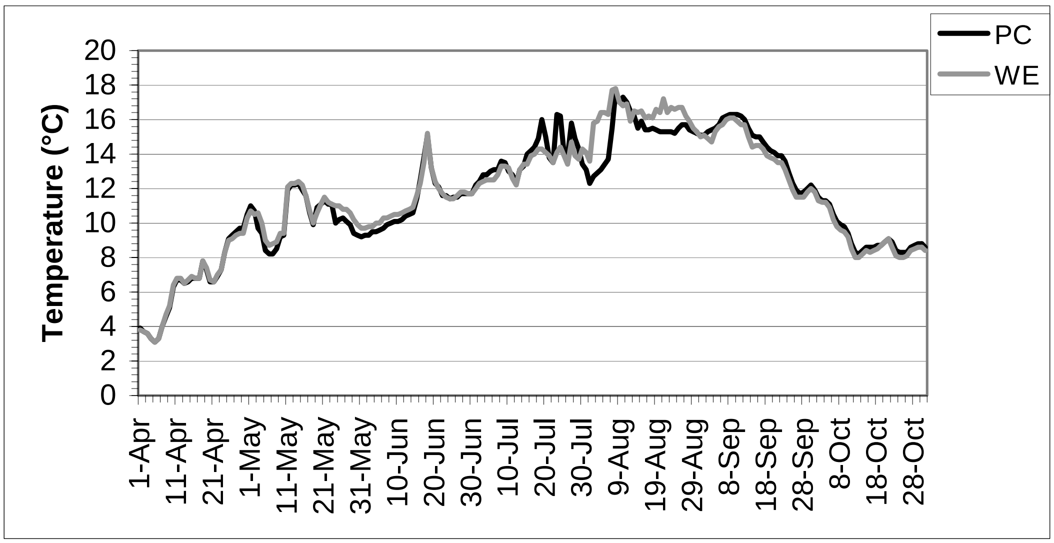


Figure 48. Simulated temperature of Fremont Canyon Powerplant bypass — 1961 conditions under operations of the Present Condition and the Water Emphasis Alternative

Figure 49 shows a time series plot of the simulated 1964 release temperatures of the Fremont Canyon bypass. Once again, for most of the year, the temperatures are similar. Noticeable differences occur during the months of August and September. Through most of August, the simulated release temperature of the Water Emphasis Alternative is as much as 2°C (3.6°F) warmer than that of the Present Condition. This is reversed in September, when the Present Condition release temperature is generally 2°C warmer. The lower temperature of the release with the Water Emphasis Alternative operation reflects the above difference in the 1964 temperature profiles.

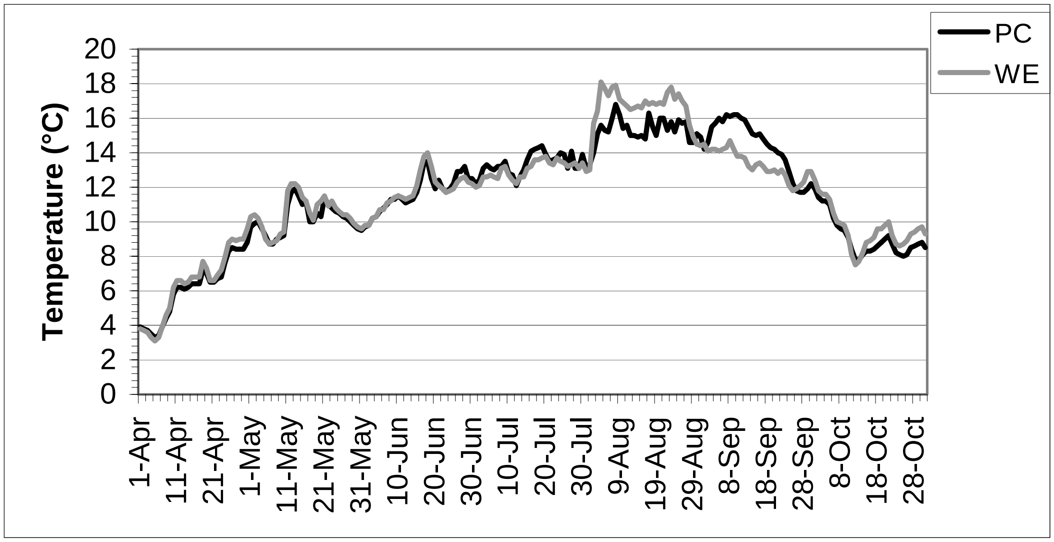


Figure 49. Simulated temperature of Fremont Canyon Powerplant bypass under 1964 conditions under operations of the Present Condition and the Water Emphasis Alternative

Effects of the Full Water Leasing Alternative – Temperature Profiles

Figure 50 shows temperature profiles form the temperature simulation of the Present Condition and the Full Water Leasing Alternative operations in 1961. The Full Water Leasing Alternative gains most of its water supply form leases that are mostly downstream from Pathfinder Reservoir. As a consequence, the reservoir retains a high pool most of the time. As is indicated by Figure 50, the total reservoir depth is greater under the Full Water Leasing Alternative operation than it would be under the Present Condition.

In the 1961 simulation, the reservoir is about 3 meters greater in total depth under the Full Water Leasing Alternative on July 21 than is the case in the Present Condition (Figure 50). The hypolimnion in the Full Water Leasing Alternative profile is somewhat larger. The metalimnion is somewhat higher in elevation under the Full Water Leasing Alternative, primarily because the reservoir content is greater. The epilimnetic and hypolimnetic temperatures are about the same, but the Present Condition profile shows stronger stratification than that of the Full Water Leasing Alternative.

By August 4, the difference in the total pool depth increases to nearly 5 meters under the Full Water Leasing Alternative operation (Figure 50). There is some temperature variation in the epilimnion of both profiles, but there is somewhat more in that of the Full Water Leasing Alternative. Interestingly, the hypolimnion of the Present Condition profile is larger than that of the Full Water Leasing Alternative, reflecting the stronger stratification in the Present Condition profile. Both profiles indicate that there might be an underflow along the bottom of the reservoir, but that is not the case. In both profiles, the inflow is in the upper part of the hypolimnion. The cold water at the bottom of the reservoir is a reflection of the isolation of cool water at the bottom of the profile as the warmer inflow moves through the reservoir above it.

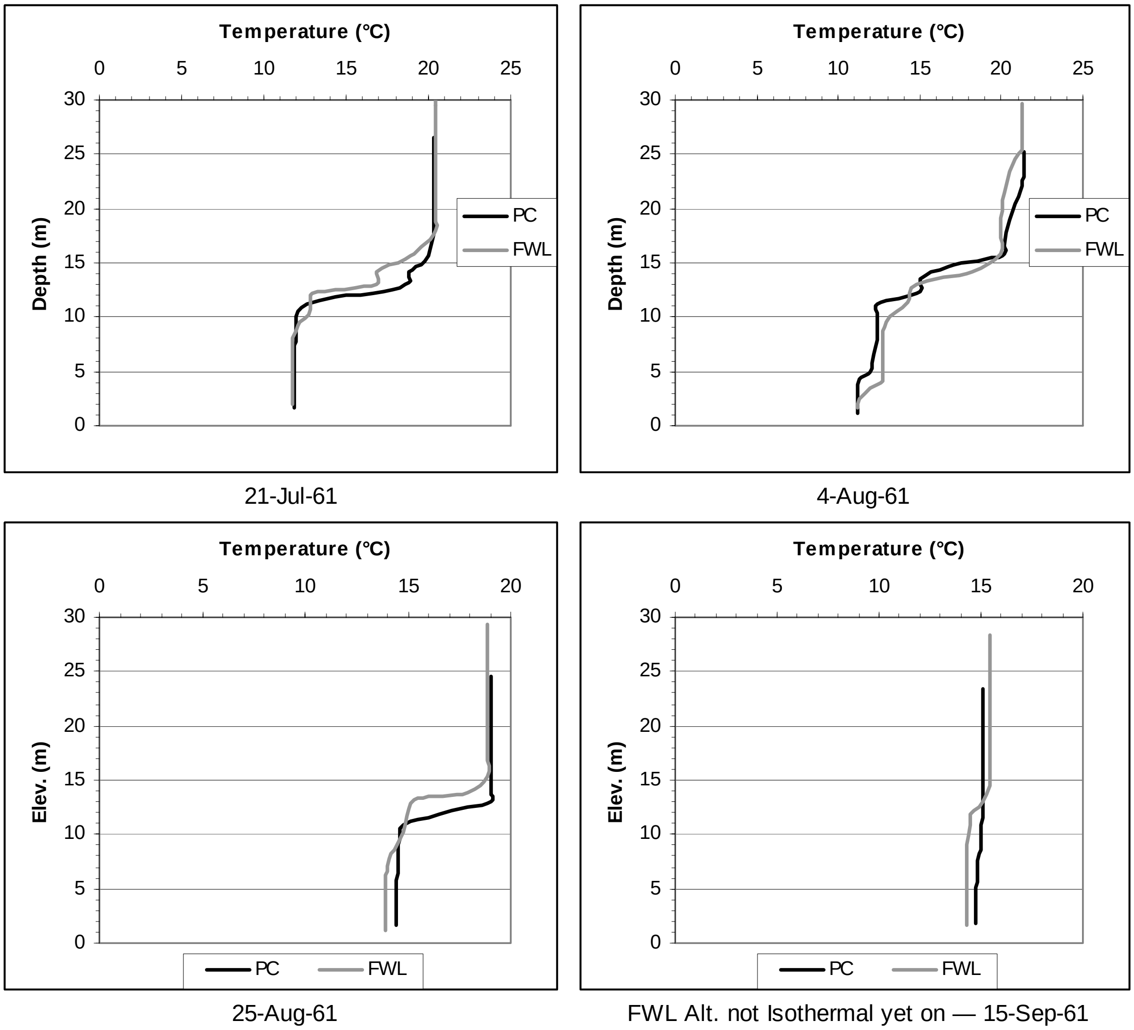


Figure 50. Comparison of selected temperature profiles in Pathfinder Reservoir in simulated 1961 conditions for the Present Condition and the Full Water Leasing alternative

The warmest epilimnetic temperatures occur on August 4. The Present Condition epilimnetic temperatures range from 21.4°C (70.5°F) at the surface to 20.1°C (68.2°F) at the top of the metalimnion. The Full Water Leasing Alternative epilimnetic temperatures range from 21.3°C (70.3°F) at the surface to 20°C (68°F) at the top of the metalimnion. Obviously, the difference is not great enough to be of any particular concern.

On August 25, the elevation difference remains at about 5 meters (16 feet). The temperature profiles are similar, but the hypolimnion under the Full Water Leasing Alternative is large, primarily reflecting the larger reservoir content. Although it may not look like it, the stratification is stronger in the Full Water Leasing Alternative profile. In both cases the epilimnetic temperatures have decreased to below 20°C (68°F).

As was noted above, the reservoir was not completely isothermal by September 15 in the Present Condition operation. It is even less so with the Full Water Leasing Alternative operation. The surface is warmer and the bottom cooler in the Full Water Leasing Alternative profile. The total temperature difference in the Full Water Leasing Alternative profile is 1.1°C (2°F), while in the Present Condition the total temperature difference is 0.4°C (0.7°F).

Figure 51 shows profiles for the Full Water Leasing Alternative and the Present Condition operations for 1964. On July 21, the epilimnetic and hypolimnetic temperatures of the 2 profiles are identical. At this point in the year, the reservoir pools have the same contents and total depths. The Present Condition profile indicates a somewhat stronger degree of stratification and a larger hypolimnion on July 21.

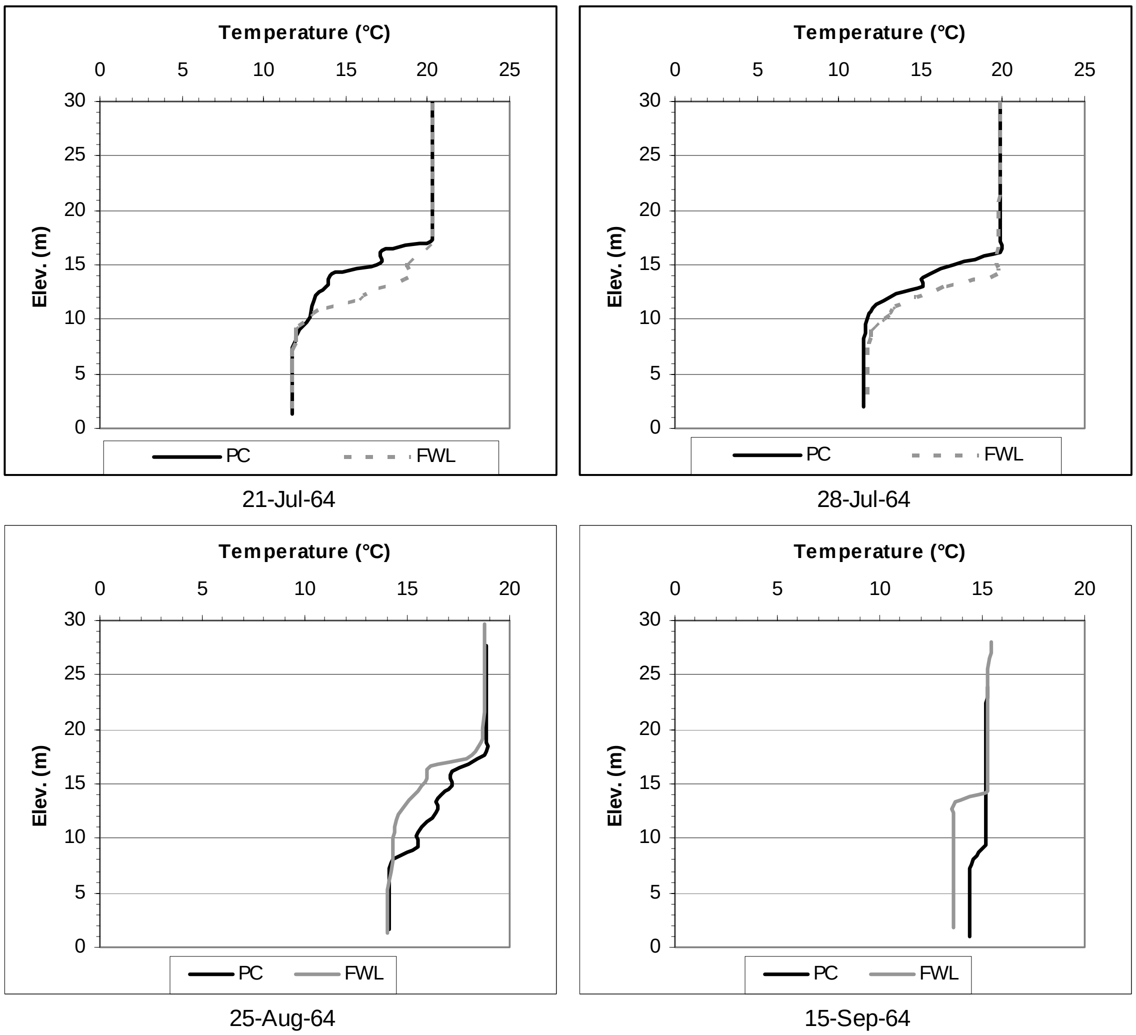


Figure 51. Comparison of selected temperature profiles in Pathfinder Reservoir in simulated 1964 conditions for the Present Condition and the Full Water Leasing alternative

The profiles for July 21 and 28 are different from all of the previous ones in that the Full Water Leasing Alternative profile is plotted with a dashed line. This makes the metalimnion of the Full Water Leasing Alternative a bit more difficult to discern, but it illustrates that the total elevation of the reservoir for both alternatives are about the same. In the case of the effects of the Full

Water Leasing Alternative, the total depth of the reservoir is the most important consideration in defining its effects.

In both sets of July 1964 profiles, the epilimnetic and hypolimnetic temperatures are overlain. The only difference in the profiles is the depth of the metalimnia. In both cases the metalimnion of the Full Water Leasing Alternative is deeper, and in both cases the size of its hypolimnion is smaller. In addition, in the Full Water Leasing Alternative July 28 profile, the epilimnion is several meters deeper.

By August 25, the characteristics of the profiles have changed. The total depth of the pool under the Full Water Leasing Alternative operation is about 2 meters greater than that of the Present Condition. The elevation of the top of the respective metalimnia of the 2 alternatives is at about the same elevation, meaning that the depth of the Full Water Leasing Alternative is greater. Stratification is somewhat stronger in the Full Water Leasing Alternative profile, and its hypolimnion is much larger.

The reservoir is not isothermal on September 14 under either operation. There is only weak stratification present with the Present Condition operation, but there is a distinct, albeit shallow (approximately 1 meter thick) thermocline present with the Full Water Leasing Alternative operation. Although the epilimnia of the 2 profiles have approximately the same temperature, the hypolimnion of the Full Water Leasing Alternative is over 1°C cooler. In general, the conditions for coldwater fish in Pathfinder Reservoir would be somewhat better with the Full Water Leasing Alternative operation than under that of the Present Condition.

Effects of the Full Water Leasing Alternative – Fremont Canyon Bypass

Figure 52 shows a time series comparison of the Fremont Canyon bypass temperatures for the Present Condition and the Full Water Leasing Alternative operations. The bypass temperature with the Full Water Leasing Alternative operation is usually cooler than that of the Present

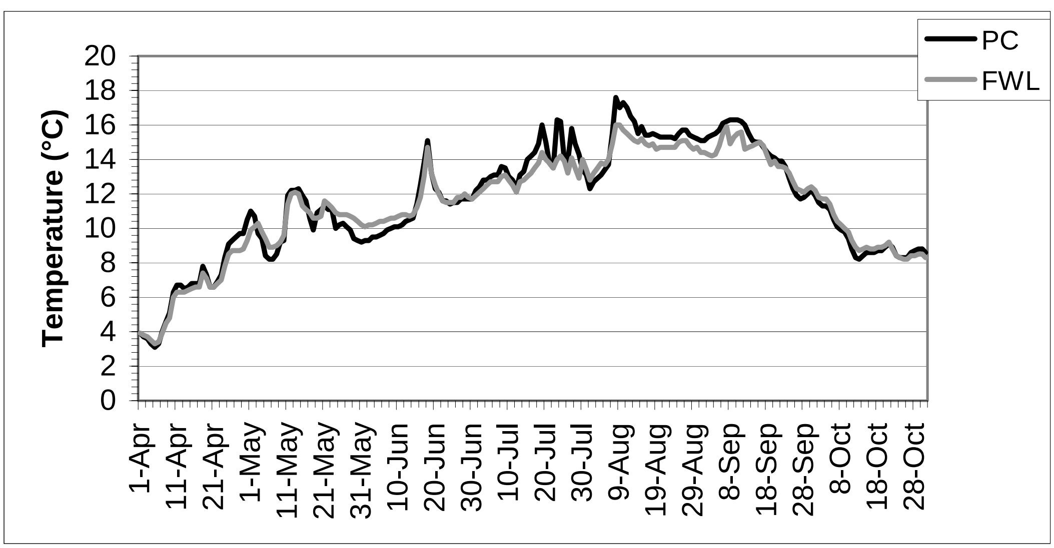


Figure 52. Simulated temperature of Fremont Canyon Powerplant bypass under 1961 conditions under operations of the Present Condition and the Full Water Leasing Alternative

Condition. Exceptions occur during 2 periods in May, when the differences are very small, and during October, when the reservoir is cooling and the temperature difference is again very small. The peak bypass temperatures in both the Present Condition and the Full Water Leasing Alternative operations occur in mid-August (Figure 52). Throughout August, the reservoir was rather strongly stratified under both operations, but the total depth was about 5 meters (16 feet) greater under the Full Water Leasing Alternative operation. The peak temperature is about 2°C lower under the Full Water Leasing Alternative, *i.e.* 16°C (61°F) *vs.* 18°C (64°F). The bypass temperature with the Full Water Leasing Alternative operation was also about 2°C lower during much of July (Figure 52).

Figure 53 shows the 1964 time series plot of the Present Condition and the Full Water Leasing Alternative operations. The interrelationships of the 1964 bypass temperatures between the 2 alternatives is much more complex than those of 1961. The complexity
is a reflection of the in-reservoir temperature relationships with total depth and the depth and strength of stratification.

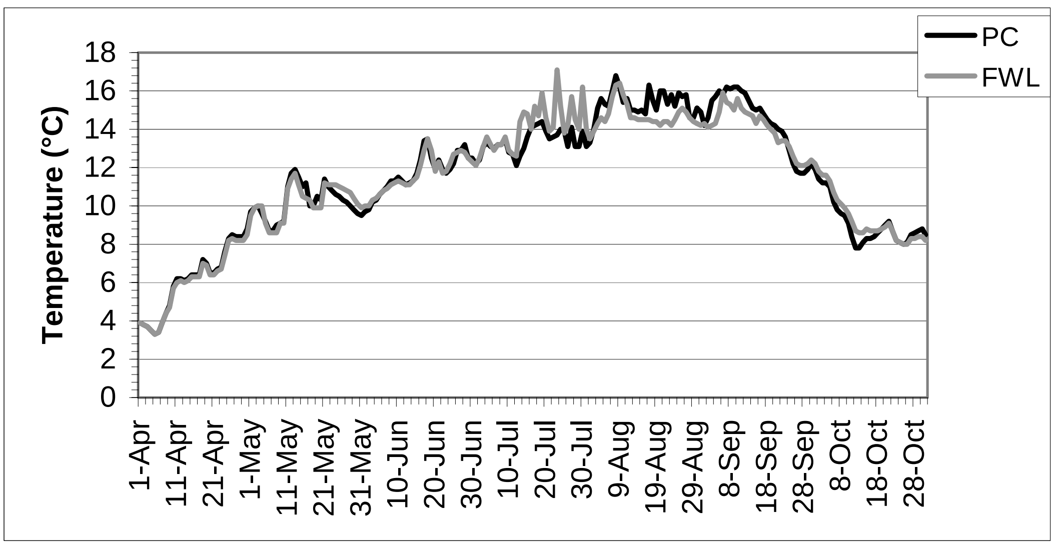


Figure 53. Simulated temperature of Fremont Canyon Powerplant bypass under 1964 conditions under operations of the Present Condition and the Full Water Leasing Alternative

The primary period in which there are temperature differences in the bypass between the 2 alternatives is July through September. During much of July, the temperature of the bypass under the Full Water Leasing Alternative operation is 2-3°C (4-5°F) warmer. In August, the temperature of the bypass with the Present Condition operation is about 2°C warmer. During much of September, the temperature of the bypass with the Present Condition operation is slightly warmer (< 1°C) than that of the Full Water Leasing Alternative operation.

The peak temperature of the bypass under the Full Water Leasing Alternative operation occurred in July (Figure 53). Under the Present Condition operation, the peak temperature of the bypass occurred in August. In something of a surprise, the peak temperature with the Full Water Leasing Alternative operation was higher than that of the Present Condition operation, *i.e.* 17.1°C and 16.3°C (62.8 and 61.3°F), respectively. In this case, the larger pool was offset by the smaller hypolimnion in the reservoir during July. With the smaller hypolimnion, withdrawals of significant amounts of water from higher in the water column allowed for an increase in the release temperature. While this may be of academic interest, from a practical perspective, even the peak release temperatures are well within the tolerances of coldwater fish.

Effects of the Wet Meadow Emphasis Alternative – Temperature Profiles

Although the Wet Meadow Alternative places more emphasis on wet meadows than the other alternatives, it still draws water from the Upper North Platte reservoirs. In some respects, its water requirements from the Upper North Platte reservoirs are greater than those of the previous alternatives. For example, the Wet Meadow Alternative draws Pathfinder Reservoir below its critical 50,000 acre-foot pool content more often than any of the other alternatives (Table 14). On this basis, the Wet Meadow Alternative has the potential for having the greatest adverse impact on Pathfinder Reservoir of any of the alternatives.

Figure 54 shows simulated temperature profiles for the Present Condition and the Wet Meadow Alternative operations for 1961 conditions. On July 21, the reservoir water surface is less than 1 meter (3 feet) lower under the Wet Meadow Alternative operation than with the Present Condition operation. The simulated epilimnetic and hypolimnetic temperatures are approximately the same under the 2 alternative operations. The metalimnia under the operations intertwine, but the stratification under the Present Condition operation is much stronger with a maximum temperature change of 6°C/m (3.3°F/ft). The maximum temperature change in the Wet Meadow Alternative profile is 2.8°C/m (1.5°F/ft).

On August 4, the water surface elevations remain less than 1 meter different under the alternative operations (Figure 54). August 4 is also the date of the maximum simulated surface temperature in Pathfinder Reservoir, but the epilimnetic temperatures under the alternative operations remain similar. However, there is a large difference in the hypolimnetic temperatures with the Wet Meadow Alternative nearly 4°C (7°F) warmer at an elevation of 10 meters (33 ft). At that point in time, most of the Wet Meadow Alternative inflow is into the metalimnion, while most of the withdrawal is from just above and the upper part of the metalimnion. These factors create currents that would account for the ragged configuration of the metalimnion in the Wet Meadow Alternative profile. Under the circumstances, the Present Condition profile exhibits the much stronger stratification, with a maximum temperature change of 3.3°C/m (1.8°F/ft). This temperature change is more than twice the maximum temperature change in the Wet Meadow Alternative profile.

On August 25, the water surface elevation with the Wet Meadow Alternative is about 2 meters (7 feet) lower than that of the Present Condition. With the Wet Meadow Alternative, the epilimnion is smaller and cooler and the metalimnion is shallower. Despite the difference in the water surface elevations, the Wet Meadow Alternative hypolimnion is much larger. The Present Condition profile once again exhibits the stronger stratification.

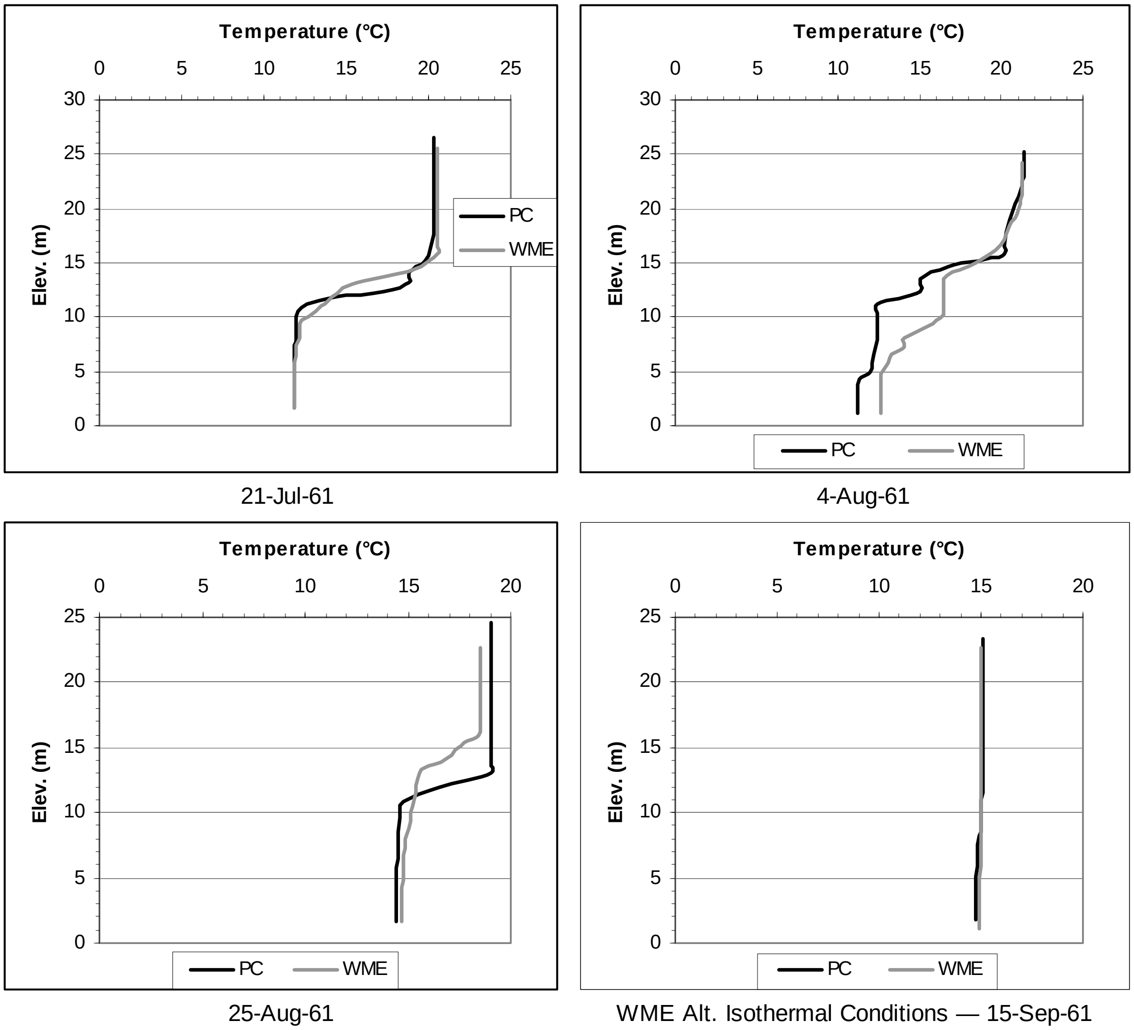


Figure 54. Comparison of selected temperature profiles in Pathfinder Reservoir in simulated 1961 conditions for the Present Condition and the Wet Meadow Emphasis alternative

By September 15, the reservoir is isothermal under the Wet Meadow Alternative operation. The reservoir is also essentially isothermal under the Present Condition operation as well. By this time, there is little difference in temperature or its distribution in the respective profiles under the alternative operations.

Figure 55 shows the 1964 temperature profiles with the Present Condition and Wet Meadow Alternative operations. The 1964 temperature profiles for the Wet Meadow Alternative look remarkably like those of the Water Emphasis Alternative, particularly in July (compare figures 34 and 42). There is also little difference between the temperature profiles of the Present Condition and the Wet Meadow Alternative operations. The only real difference is that the Wet Meadow Alternative epilimnion is slightly warmer (0.3°C [0.5°F]).

By August 25, the reservoir is approaching its minimum operating level under the Wet Meadow Alternative operation. The water surface elevation is nearly 10 meters (33 feet) lower under the

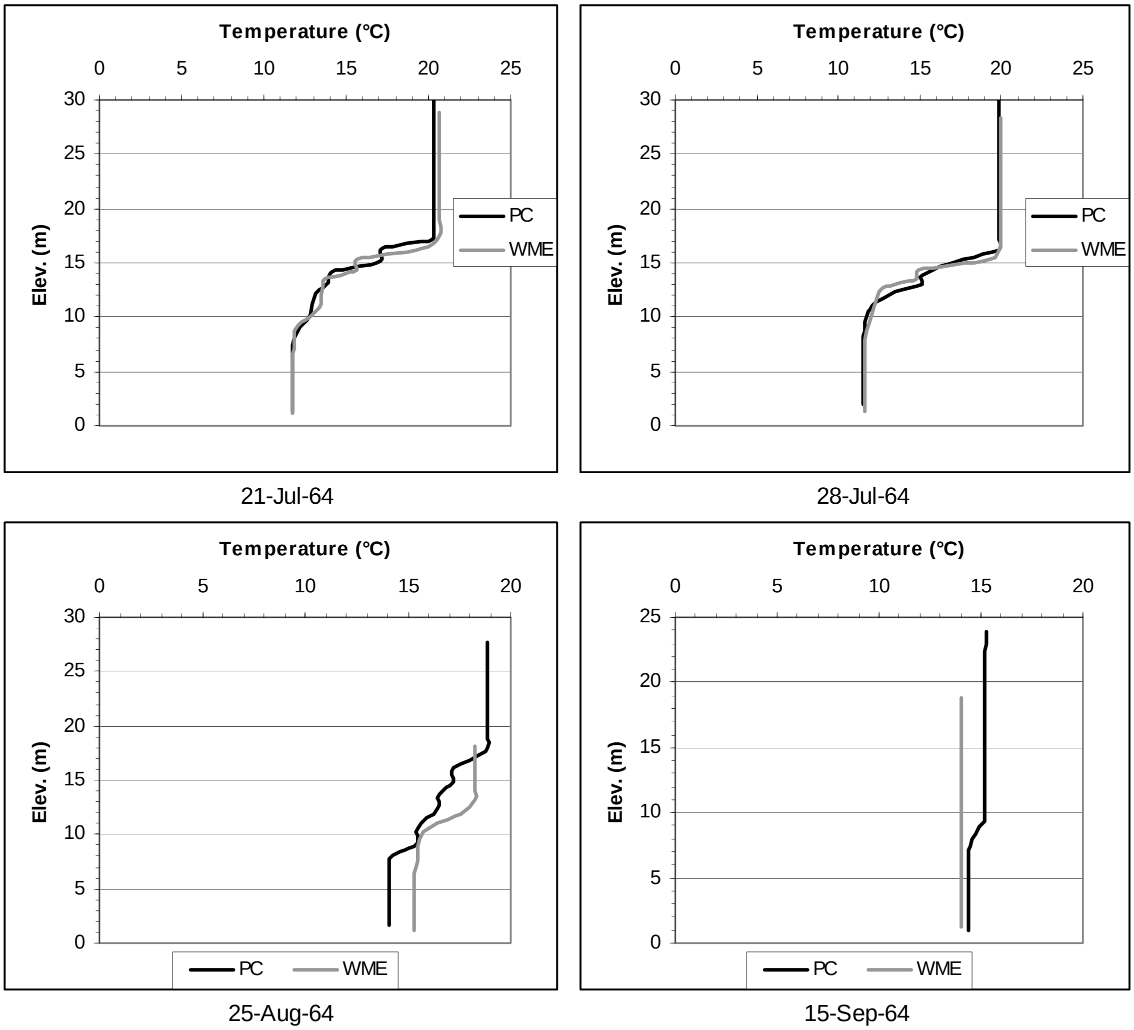


Figure 55. Comparison of selected temperature profiles in Pathfinder Reservoir in simulated 1964 conditions for the Present Condition and the Wet Meadow Emphasis alternative

Wet Meadow Alternative operation. Despite the disparity in their respective water surface elevations, the Wet Meadow Alternative has the larger hypolimnion, reflecting the stronger stratification in its profile. The hypolimnetic temperature of the Wet Meadow Alternative profile is more than 1°C warmer, while its epilimnetic temperature is about 0.5°C cooler than that of the Present Condition. In this case, the lower water surface elevation and associated smaller reservoir pool lead to a more compact temperature profile.

By September 15, the reservoir is isothermal under the Wet Meadow Alternative operation. The reservoir is also cooler; the entire Wet Meadow Alternative profile is to the left of the Present Condition profile on Figure 55. Once again, the smaller reservoir pool is cooling more rapidly than the larger one of the Present Condition operation.

Effects of the Wet Meadow Emphasis Alternative – Fremont Canyon Bypass

Figure 56 shows the 1961 time series plot of the temperatures of the Fremont Canyon bypass with the Present Condition and the Wet Meadow Alternative operations. The release temperatures of the 2 alternative operations are similar throughout the year. Differences are small. The only unexpected result is that the maximum temperature occurs under the Present Condition operation. In the spring, the temperature of the bypass is slightly warmer under the Wet Meadow Alternative operation, although by less than 1°C. Through the summer, the higher release temperature fluctuates between the 2 alternative operations. The Present Condition operation shows the greater degree of fluctuation in the temperature of the bypass during the summer (figure 56).

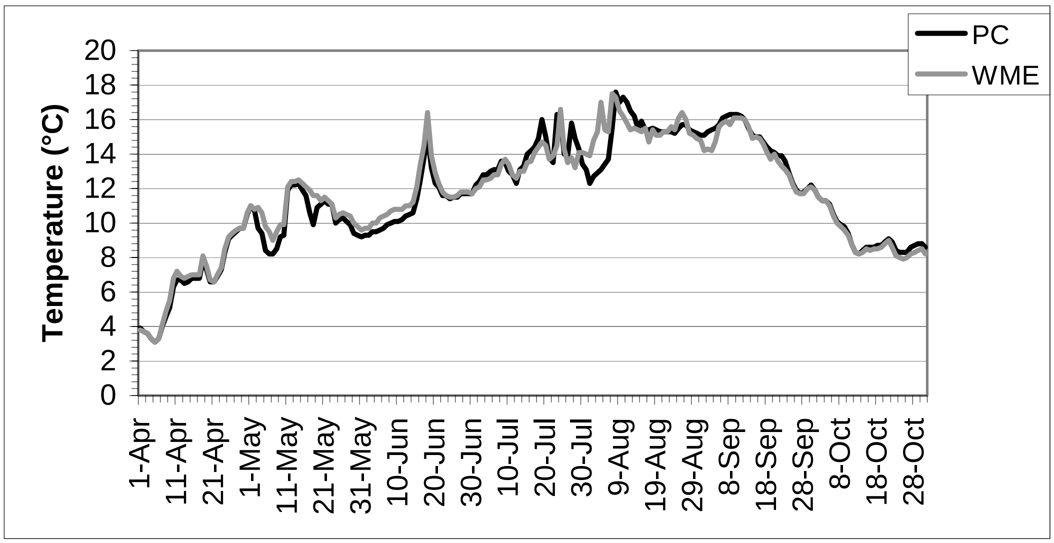


Figure 56. Simulated temperature of Fremont Canyon Powerplant bypass under 1961 conditions under operations of the Present Condition and the Wet Meadow Emphasis Alternative

Figure 57 shows the 1964 time series plot of the temperature of the Fremont Canyon bypass for the Present Condition and the Wet Meadow Alternative operations. During the spring under the 1964 conditions, the temperature of the bypass under the Wet Meadow Alternative operation is slightly warmer than that of the Present Condition. This relationship is reversed in the early summer when the temperature of the bypass is slightly warmer under the Present Condition operation. Throughout most of August, the relationship between the release temperatures is once again reversed and that of the Wet Meadow Alternative is noticeably warmer (Figure 57). In September, the relationship is once again reversed and the temperature of the bypass under the Present Condition operation is warmest. Of note once again is that the maximum temperature of the release under either operation is around 18°C (64°F) and well within the tolerance of coldwater fish.

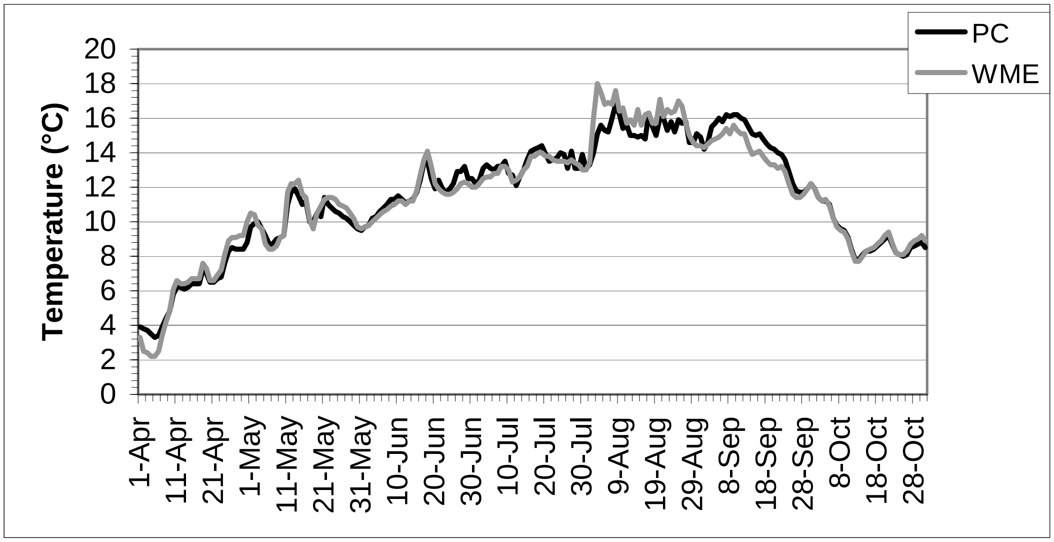


Figure 57. Simulated temperature of Fremont Canyon Powerplant bypass under 1964 conditions under operations of the Present Condition and the Wet Meadow Emphasis Alternative

Summary of Temperature Model Results

Based on the temperature simulations of the various alternative operations, there are a number of effects on the temperature profiles in Pathfinder Reservoir. However, there are no great effects on maximum or minimum temperatures. This modeling effort was undertaken based on the hypothesis that the smaller reservoir pools that would result from some of the alternatives would have higher temperatures and have an adverse effect on coldwater fish. This does not appear to be the case, at least not strictly due to higher than suitable temperatures in and of themselves.

These results are in many respects consistent with a study conducted by Leopold (2000) in a study of natural lakes in the Wind River Mountains north of the North Platte River. In that study, he found that the bottom temperatures were independent of lake bathymetry. He further observed that large lakes have a larger mixed layer (epilimnion) than smaller lakes. The net effect was that heat budget is controlled more by climate than the size or morphometry of a lake. This appears to be the case with the Upper North Platte reservoirs. Although the operations do appear to affect the temperature profiles, primarily by the relationship between inflows, the depth and thickness of the metalimnion, and the withdrawal zone, there is no overall effect on the reservoir heat budget. As a consequence, temperatures remain suitable for coldwater fish.

The effect of climate is probably best illustrated by Figure 28. In a relatively free flowing river, the Sweetwater River, the peak temperature occurs in July. After that, cooling begins. Critical conditions would be expected to occur in a reservoir in August. However, in the Upper North Platte reservoirs, cooling begins prior to that and even the epilimnion becomes suitable for coldwater fish.

Dissolved Oxygen Depletion

The probability of dissolved oxygen (DO) depletion in Pathfinder Reservoir was estimated using empirical models that are presented in Reckhow and Chapra (1983). The models are based on nutrient (total phosphorus) loadings. However, in order to estimate the total phosphorus in the inflow to Pathfinder Reservoir, some of the same models had to be applied to Seminoe Reservoir. It was a simple matter to apply the same DO models used for Pathfinder Reservoir to Seminoe Reservoir. The results for both Seminoe and Pathfinder reservoirs will be presented here for the Present Condition and each of the 4 alternatives.

Methods

Nutrient data for sites on the North Platte, Medicine Bow, and Sweetwater rivers upstream from Seminoe and Pathfinder reservoirs were acquired from the USGS National Water Information System (NWIS) website. Relationships to flow and other constituents with a more extensive record, *e.g.* EC and TDS, were explored. Total phosphorus is composed of both dissolved and particulate fractions. Each of the fractions tend to relate differently to flow. The dissolved fraction, like TDS or EC in the Sweetwater River, tends to be dominated by dilution and should show an inverse relationship to flow. Alternatively, particulate phosphorus, like the TSS in the Sweetwater River, tends to be dominated by transport and relates positively to flow, particularly if erosion is the primary phosphorus source. However, the particulate fraction (and to some extent, the dissolved fraction as well) is usually composed of organic and inorganic fractions. The inorganic fraction usually dominates in erosive sources, while the organic fraction tends to originate within a water body. The problem with this is that the competing mechanisms tend to cancel each other out unless one form or the other is overwhelmingly dominant. In the data sets retrieved from NWIS, there are no clear-cut relationships between total phosphorus and flow ($r^2 \leq 0.1$) that would allow for the development of regression relationships like those for TDS or EC in the Sweetwater River that could be used for the estimation of concentrations in the Seminoe and Pathfinder reservoir inflows. More complex relationships like those for the TDS in the North Platte River upstream from Pathfinder Reservoir were explored.

Multiple regressions of the total phosphorus concentration on a variety of independent variables were developed. None of these had particularly good R^2-values either, *i.e.* the best were between 0.2 and 0.3 (Table 17). However, the nutrient loading models are based on annual loads to the reservoir. On that basis, the predictive capability of the models was evaluated against the median of the observed data. Those results showed good agreement (Table 17). The multiple regressions also did a reasonable job of predicting the minimum values, which consisted of values that were assigned to results reported as less than a reporting limit of 0.01 mg/L. Neither regression did a good job of predicting the maximum values. On the basis of the agreement between the observed and predicted median total phosphorus concentrations, the multiple regressions were used to estimate the inflow total phosphorus to Seminoe Reservoir.

The total phosphorus data on the North Platte River upstream from Pathfinder Reservoir were very limited. The data consisted of 12 samples collected during water years 1988 and 1989. To make matters worse, 2 of the samples had results that were less than a detection limit that was

Table 17. Summary of the North Platte and Medicine Bow multiple regressions – dependent variable is log total phosphorus				
	North Platte Regression		Medicine Bow Regression	
Overall	R^2	0.228	R^2	0.268
	Samples	195	Samples	128
Components	Variables	Coefficient	Variables	Coefficient
	Constant	-27.812888	Constant	-10.022744
	ln flow	0.665592	Flow	0.000320
	ln EC	4.002268	Ann. Flow	-0.005295
	EC	-0.009575	log flow	0.880206
			log EC	2.345982
Performance	North Platte River		Medicine Bow River	
	Predicted	Observed	Predicted	Observed
Minimum	0.007	0.004	0.007	0.004
Median	0.029	0.030	0.029	0.030
Maximum	0.227	0.740	0.426	2.200

greater than almost ½ the reported results, such that the median total phosphorus concentration is one of the less than values (< 0.03 mg/L). In addition, the best regression of total phosphorus on flow, using log-transformed data, on had an r^2 of 0.008. Because of all of the problems with the North Platte total phosphorus data, the inflow to Pathfinder Reservoir was based on the Seminoe Reservoir predicted concentration.

The relationship between the total phosphorus in the Sweetwater River and other variables was also poor. The best correlation for the total phosphorus concentration in the Sweetwater River was with date (r = 0.2), followed closely by one with water year (r = 0.19). Neither is a usable predictor in that the water years in the data record are much more limited than those in the operations study used to compare alternatives, although both correlations indicate a somewhat long-term trend in the data. However, that result does not help in developing a predictor for total phosphorus.

In exploring the Sweetwater River total phosphorus data further, there appeared to be a seasonal pattern. The seasonal pattern was strongest in the higher total phosphorus concentrations, but there was still too much scatter in the data to develop a good relationship. In the process of analyzing the data, the medians and means of the higher and lower concentrations were plotted. Regressions of the median total phosphorus concentrations were developed. These regressions and the median total phosphorus concentrations are shown on Figure 58. The split in the total phosphorus concentration that is used for Figure 58 is greater or less than 0.05 mg/L. The problem is that the regressions shown on Figure 58 do not provide a method for predicting total phosphorus concentrations over the longer term record. For the regressions to be usable, a method for predicting when the total phosphorus concentration is in one or the other of the 2 classes is needed.

Discriminant analysis was used to develop multivariate classification functions to predict whether the total phosphorus concentration should be less than 0.05 mg/L or greater than or equal to 0.05 mg/L. As was done to derive the multiple regressions described above, a variety of variables were entered into a stepwise discriminant analysis to develop the best predictor. The final results are shown in Table 18. The classification functions are essentially a pair of multiple regression equations that calculate values that are compared against each other. The observation is assigned to one set or the other depending on which classification function is greater. In this case, the different quadratic regressions were applied on that basis, and a total phosphorus concentration was calculated for each month in the data set in the North Platte operations model. The geometric mean total phosphorus for each year was used in calculating the annual total phosphorus load to Pathfinder Reservoir. The overall estimated geometric mean total phosphorus concentration in the Sweetwater River upstream from Pathfinder Reservoir was 0.028 mg/L.

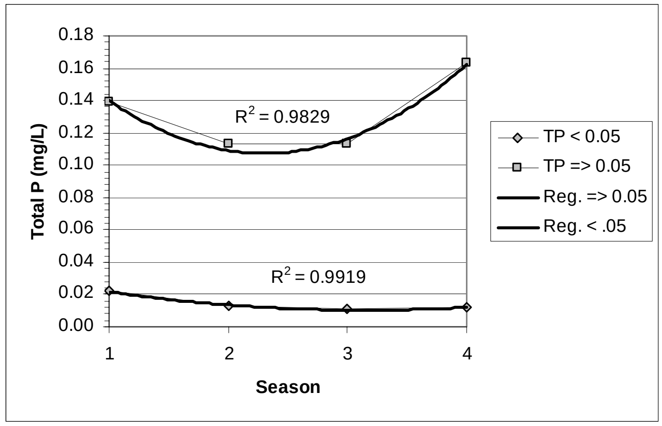


Figure 58. Quadratic regressions of median total phosphorus concentrations on season

Table 18. Discriminant Analysis Results for Calculating Total Phosphorus (TP) Concentrations in the Sweetwater River upstream from Pathfinder Reservoir				
Variables	Classification functions		Group means	
	TP < 0.05	TP ≥ 0.05	TP < 0.05	TP ≥ 0.05
Ave. Ann. Flow (ft^3/s)	0.092	0.115859	72.8	86.4
Season[1]	-3.234	-3.8847	2.7	2.0
ln Flow (ft^3/s)	170.154	171.1697	4.191127	4.793957
ln EC (µS/cm)	823.275	827.6623	6.053884	5.962378
Constant	-2847.816	-2880.17	—	—
[1] Seasons: 1 = spring…4 = winter				

The total phosphorus driven relationships for predicting the oxic/anoxic probabilities in the late summer are based on the areal total phosphorus load. All of the data used in computing the various parameters in the empirical model are in metric units. The first activity was to convert all of the necessary inputs from the North Platte EIS model output into metric units. The needed variables include the reservoir inflow, the average end-of-month reservoir contents, the reservoir mean depth, and the area. The areal total phosphorus load is calculated as the inflow total phosphorus load (Kg/year) divided by the area in m^2.

The probability of the reservoir being oxic in the hypolimnion in the late summer is calculated from a discriminant function similar to that described above; however, there are 3 possible categories into which an observation, *i.e.* year, can be assigned. These include oxic, indeterminate, but probably oxic, and anoxic. The anoxic category was based on whether there was an observation within a DO profile that showed anoxic conditions. In other words, there may or may not be complete anoxia in the hypolimnion (Reckhow and Chapra, 1983). The discriminant analysis was performed on data included in the EPA’s National Eutrophication Survey (NES) database. It should be noted that Seminoe Reservoir was included in the NES.

The discriminant function (d.f.) is calculated from the following formula:

$$\text{d.f.} = 10^{4.68}L^{1.95} / q_s^{1.85}z^{2.58}$$

where: L = the areal total phosphorus load (g/m^2/year),

z = the reservoir mean depth [= contents (m^3)/area (m^2)]in meters

q_s = the mean depth divided by the hydraulic residence time (m/year).

The hydraulic residence time is the average reservoir contents divided by the inflow for the year. For classification purposes, values of d.f. less than 0.2 indicate oxic conditions, while values greater than 0.6 indicate anoxic conditions. Intermediate values are considered indeterminate, but probably oxic (*ibid*.).

Reckhow and Chapra (1983) also present an equation to calculate the probability that the late summer hypolimnion will be oxic. Obviously, the complement of that probability is the probability that the late summer hypolimnion will be anoxic. The following equation is used to calculate the oxic probability:

$$P_{oxic} = 1 / (1 + (10^5 z^{2.49} L^2 q_s^{1.78})),$$

where P_{oxic} is the probability of a late summer oxic hypolimnion. The other variables in the equation are defined above. The equation is a logistic regression based on log-transformed data, in which the exponents are the coefficients of the regression.

As was noted above, the average reservoir total phosphorus concentration can also be estimated from the areal total phosphorus load. The equation for the average total phosphorus from Reckhow and Chapra (1983) is:

$$P = L / (11.4 + 1.4q_s),$$

where P is the average total phosphorus concentration in the reservoir. The other 2 variables are defined above. The variable, q_s can also be used as an estimator of the phosphorus settling rate in the reservoir. Consequently, the above equation also accounts for total phosphorus settling as it passes through a reservoir. In their review of the derivation of the equation, Reckhow and Chapra (1983) indicate that it integrates the areal total phosphorus load over time and space to estimate the annual average total phosphorus concentration in a reservoir. The relationship to the outflow also assumes that $P = P_{out}$, which over the course of a year is probably not unreasonable. The above equation was applied to the Seminoe Reservoir results to estimate the inflow total phosphorus concentration in the North Platte River upstream from Pathfinder Reservoir.

Results

The main purpose for applying the nutrient loading models to Seminoe Reservoir was to estimate the total phosphorus concentration in the outflow, and thus the inflow to Pathfinder Reservoir. As was described elsewhere, Sartoris *et al.* (1981) studied Seminoe and Pathfinder reservoirs in the late 1970's. Although they reported nutrient results, the only phosphorus results were for dissolved orthophosphate, which is only 1 component of the inorganic fraction of total phosphorus and cannot be used as a basis for comparison with the estimates developed here. As part of the National Eutrophication Survey (NES), EPA (1977) studied Seminoe Reservoir in 1975. EPA (1977) reported monthly results for total phosphorus in the Seminoe Dam releases for the water year 1975. Those results showed a range of total phosphorus concentrations from less than 0.010 mg/L to 0.080 mg/L, with an average of 0.024 mg/L. The average total phosphorus concentration in the Seminoe Dam releases that was estimated from the loadings in the Present Condition was also 0.024 mg/L, with a range from 0.011 to 0.040 mg/L. An overall comparison is illustrated on Figure 59, which shows a time series of each of the estimated annual averages and the EPA average as a constant.

The range in the EPA (1977) data set is for monthly measurements within 1 water year, while the range in the estimates developed from the empirical nutrient loading model are all annual averages. The estimated average total phosphorus concentration in 1975 from the loading model is 0.015 mg/L; the EPA data indicate that the average in that year was 0.024. However, the Present Condition is a simulation and not exactly identical to the historical condition. In addition, EPA (1977) indicates that some of the other tributaries to Seminoe contribute measurably to the nutrient load of Seminoe Reservoir. The loading model only considers the North Platte and Medicine Bow rivers. If the other tributaries do contribute significant phosphorus loads, the empirical model underestimates the actual outflow total phosphorus concentration. Despite all of this, the comparison on Figure 59 does indicate that the empirical model results are in the proverbial “ball park”, which is about all that can be hoped for in this type of analysis.

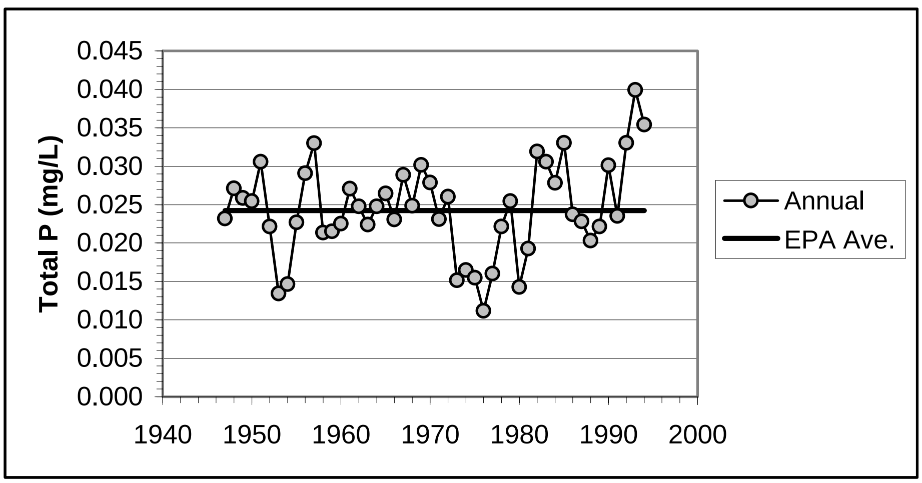


Figure 59. Comparison of the estimated annual average total phosphorus concentration in the Seminoe Dam release with that from EPA (1977)

The original areal nutrient loading concept was developed using natural lakes. Natural lakes do not fluctuate widely in volume, mainly because the inflows and outflows are at the surface. If the water level falls, there will be no outlet, a condition that is likely only in extreme drought or in seepage lakes. In the applying the concept to reservoirs, EPA attempted to normalize the inflows to make the loading reflect a wide range of flow conditions. However, the second component of the areal load reflected only the conditions during the year that the samples were collected. Reservoirs, unlike natural lakes, may fluctuate greatly in terms of both area and content. In terms of the parameters of the empirical model described above, a reservoir in different years may be like different lakes in the same year. Consequently, it does seem not unreasonable to apply the results on a year-by-year basis to evaluate a wide range of flows, volumes, depths, and areas of a reservoir and to look at the results of the model on that basis.

Oxic Conditions Comparison of Alternatives in Seminoe Reservoir

A discriminant function (d.f.) was calculated for each year in the record used in the North Platte EIS model (1947-94) for the Present Condition and each of the 4 alternatives. The results are summarized in Table 19. For the Present Condition and each of the 4 alternatives, the greatest percentage of the d.f. for Seminoe Reservoir fall in the indeterminate category. The results for 2 of the 4 alternatives and the Present Condition show a d.f. in the anoxic category less than 5 percent of the time, while the results in that category for the Full Water Leasing and Wet Meadow Emphasis alternatives are between 8 and 10 percent of the time.

Table 19. Frequency (%) of discriminant function levels and mean probability of a late-summer oxic hypolimnion in Seminoe Reservoir under the Present Condition and each of the action alternatives

Condition/Alternative	% Oxic (d.f. < 0.2)	% Indeterminate (0.2 < d.f. <0.6)	% Anoxic (d.f. > 0.6)	Probability oxic
Present condition	35	63	2	0.586
Governance Committee	33	63	4	0.574
Water Emphasis	35	61	4	0.565
Full Water Leasing	29	61	10	0.551
Wet Meadow Emphasis	29	63	8	0.553

The last column of Table 19 shows the arithmetic mean probability of an oxic late-summer hypolimnion in Seminoe Reservoir. All of the probabilities are around 0.6. The probability of an oxic late-summer hypolimnion is slightly higher under the Present Condition than for any of the alternatives. The Governance Committee alternative has the highest probability of any of the action alternatives.

Figure 60 shows a time series plot of the d.f. for the Present Condition and the Governance Committee alternative. In most of the years, the Governance Committee alternative and the Present Condition d.f. are overlain. Figure 3 also shows the boundaries of the oxic and anoxic d.f. areas of the plot. As was noted in Table 19, most of the d.f. are in the intermediate category. This is well illustrated on Figure 60 by the data points that plot between the boundaries. Most of the remaining annual d.f. plot below the oxic boundary,
indicating a high probability that late summer oxic conditions should prevail in those years.

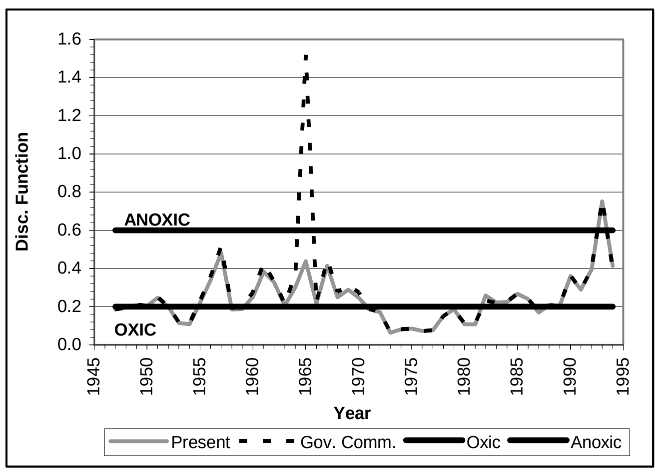


Figure 60. Annual discriminant functions for the Present Condition and the Governance Committee alternative in Seminoe Reservoir

There are 2 years in which the d.f. of the Governance Committee alternative and 1 year in which the d.f. of the Present Condition exceed the anoxic boundary and indicate a high probability that a late summer anoxic hypolimnion will develop. In 1993, the d.f. is near 0.8 for both the Present Condition and the Governance Committee alternative. Although conditions for fish would seem to be adverse in Seminoe Reservoir with that result, it would not be considered an adverse impact of the Program, as there is no change from the Present Condition. Alternatively, in 1965, the d.f. for the Governance Committee alternative is over 3 times that of the Present Condition. The associated probability of anoxia for a d.f. as large as the one for the Governance Committee alternative is in excess of 0.8. The equivalent probability for the Present Condition would be slightly less than 0.6.

The Present Condition is a simulated representation of current North Platte Basin water operations. This is not identical to an historical condition, but it should be reasonably close. If so, then historical data should provide some insight into the frequency of hypolimnetic anoxia in Seminoe Reservoir. Figure 61 shows a time series plot of the bottom DO data from Seminoe Reservoir from monitoring and studies conducted in the reservoir between 1973 and 1983 by personnel from the USGS and Reclamation. The plot also shows the DO concentration at which the water would be 100 percent saturated with oxygen. The difference in the 2 concentrations is the degree of DO depletion.

The data on Figure 61 appear to show 2 years in which the DO fell to 0 mg/L, *i.e.* anoxic conditions as defined in the derivation of the oxic d.f. However, the data point in August 1978, which appears to indicate anoxia, was actually a measurement of 0.15 mg/L. In the EPA NES data base, this would have appeared as 0.2 mg/L, because the data were reported to 1 decimal place accuracy and would not have represented anoxic conditions as defined by Reckhow and Chapra (1983). The EPA NES data base consisted of a set of samples collected within the course of 1 year for each sampled lake or reservoir. There are 10 years in which DO measurements were made in Seminoe Reservoir and included on Figure 61. On this basis, the frequency of anoxia in Seminoe Reservoir is 1 in 10 or 0.1. This could also be considered the probability of anoxia if the data included on Figure 4 are representative of longer-term conditions in the reservoir.

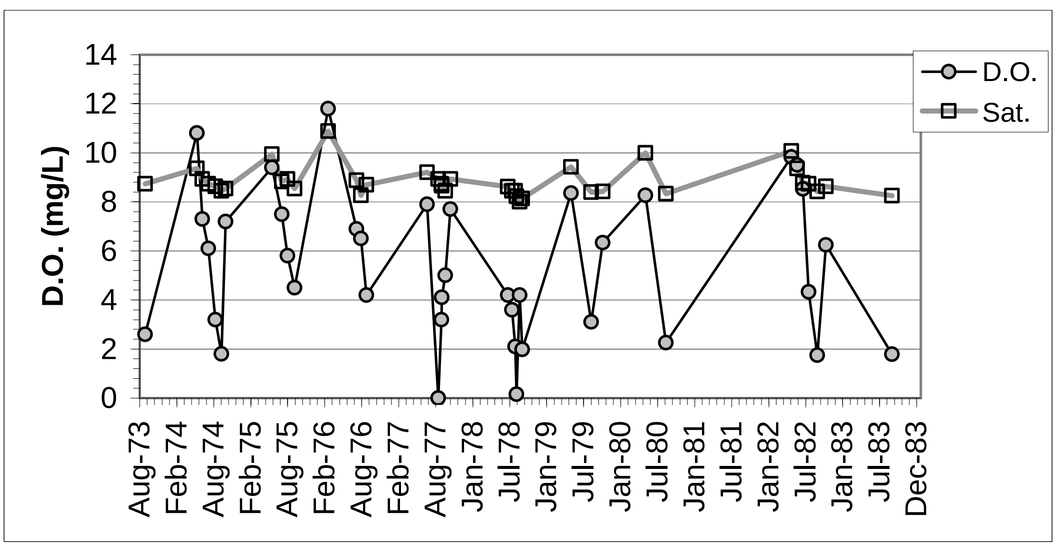


Figure 61. Bottom DO and saturation DO concentrations in Seminoe Reservoir

Several factors should be considered in interpreting the data on Figure 61 and the oxic d.f. The minimum DO in most years (6 of 10) occurs in mid- to late August, but there are 4 years in which the minimum was observed in September. In 2 of the 4 years in which the minimum was observed in September, there was no measurement in August. This raises the question as to whether the minimum in the data set for those years is the actual minimum for the year. Alternatively, in the other 2 years in which the minimum occurred in September, there was a higher DO measurement in August, indicating that DO depletion continued beyond August. To further complicate matters, the anoxic measurement was made in mid-August 1977. There were 2 other DO measurements later in August 1977 that showed progressively higher DO concentrations. In other words, the actual minimum DO is likely to have been missed in some (or most?) years. In all of the Seminoe measured profiles, the minimum DO occurred in the deepest measurement in the profile. In some cases, the bottom of the measured profile was well above the bottom of the reservoir at the reported elevation of the pool. In these cases, the actual minimum DO is also likely to have been missed in that set of samples. All of this points to the fact that there is uncertainty in the data for anoxia in Seminoe Reservoir.

EPA (1977) reported the measured profiles from Seminoe Reservoir. The minimum reported DO at the site near the dam (the one plotted on Figure 61) was 5.4 mg/L on August 27, 1975. Note that the EPA NES data are not included in the data set plotted on Figure 61. The USGS reported a bottom DO concentration of 5.8 mg/L on August 14 and 4.5 mg/L on September 16, 1977. This latter measurement indicates that the EPA NES data set does not include the minimum for 1975. Furthermore, the 4.5 mg/L DO measurement was one of the higher annual minimum concentrations in the data set plotted on Figure 61. This result indicates that the EPA NES data were based on a somewhat atypical year, at least as far as the DO in Seminoe Reservoir is concerned. It should also be noted that the EPA NES database includes data on lakes and reservoirs from across the country collected over a period of several years and such a phenomenon as shown in the Seminoe data may be damped out when the whole database is considered.

Figure 62 shows a plot based on data sets similar to those of Figure 3, but comparing the d.f. of the Present Condition and the Water Emphasis alternative. Figure 62 looks very much like Figure 60 in terms of the results, but a closer look at Figure 62 reveals differences as well. Where the d.f. of the Present Condition and the Governance Committee alternative appeared to be overlain in most years, there is more divergence between the d.f. of the Present Condition and the Water Emphasis alternative, particularly during the period 1955 through 1970. In all years in that period, there is an increase in the d.f. with the Water Emphasis alternative and by inference a decrease in the

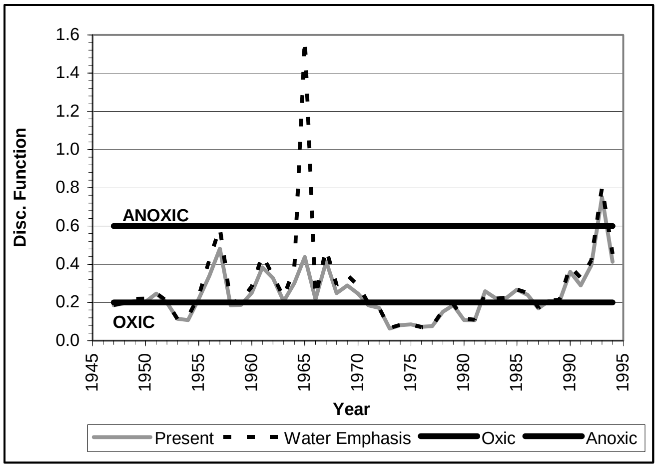


Figure 62. Annual discriminant functions for the Present Condition and the Water Emphasis alternative in Seminoe Reservoir

probability of a late summer oxic hypolimnion. As was the case for the Governance Committee alternative comparison to the Present Condition in 1993, there is no difference for the Water Emphasis alternative. Also like the Governance Committee alternative comparison to the Present Condition, there is a decrease in the probability of a late summer oxic hypolimnion in 1965 with the Water Emphasis alternative. In this case the results are again similar between the Governance Committee alternative and the Water Emphasis alternative comparisons to the Present Condition. The 1965 probability associated with the Water Emphasis alternative is 0.16, like that of the Governance Committee alternative. In terms of the oxygen regime in Seminoe Reservoir and based on Table 19, the overall effect of the Water Emphasis alternative is little different from that of the Governance Committee alternative.

Figure 63 shows a comparison between the d.f. of the Present Condition and the Full Water Leasing alternative. The first thing of note on Figure 63 in comparison with figures 47 and 48 is the magnitude of the ordinate. On figures 47 and 49, the upper limit on the ordinate is 1.6; the upper limit on Figure 6 is 1.0, or about ⅔ of the previous comparisons. This result only means that the maximum d.f. of the Full Water Leasing alternative is much smaller than was the case for the previous 2 alternatives. It also means that the probability of a late summer oxic hypolimnion is correspondingly greater in that 1 year. However, like the Water Emphasis alternative, the Full Water Leasing alternative shows a d.f. greater than those of the Present Condition through the period 1955 though 1970. The maximum d.f. for the Full Water Leasing alternative occurs under the 1993 conditions, rather than the 1965 conditions, which year now ranks 2nd highest, but still shows an increase relative to that of the Present Condition. Based on the data in Table 15, the overall probability of a late-summer oxic hypolimnion is the lowest of any of the alternatives and would indicate the most severe impact. However, this is probably not the case. Although the average probability of a late-summer oxic hypolimnion is lower for the Full Water Leasing alternative than for the other alternatives, its range is much smaller. The difference in the range is at the lower end; the maximum probabilities for all alternatives is 0.83. Where with the other alternatives, the occurrence of anoxia in the hypolimnion is a virtual certainty in some years, under the Full Water Leasing alternative, there is a somewhat intermediate probability of occurrence in those years.

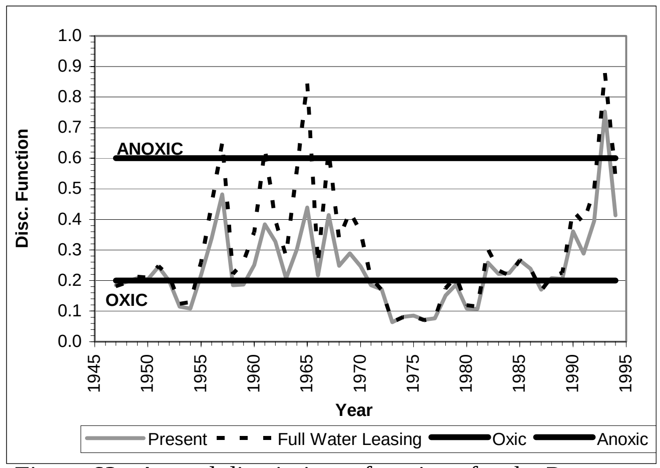


Figure 63. Annual discriminant functions for the Present Condition and the Full Water Leasing alternative in Seminoe Reservoir

Figure 64 shows a comparison between the Present Condition and the Wet Meadow Emphasis alternative. The ordinate on Figure 64 has a maximum d.f. of 1.8, which is the highest of any of the action alternatives. The maximum is d.f. of the Wet Meadow Emphasis alternative is actually only slightly greater than those of the Governance Committee and Water Emphasis alternatives. The d.f. of the Wet Meadow Emphasis alternative is once again noticeably greater than those of the Present Condition through the period 1955 through 1970 and in the 1990s.

However, the overall probability of a late-summer oxic hypolimnion is vary near that of the Full Water Leasing alternative (Table 19), all of which shows that the effect of 1 year is not a good barometer of the overall effect.

The primary reason for conducting the more intensive analysis of the effects of Program alternatives on the North Platte Reservoirs had to do with the effects on operations during the period 1961 through 1965, with emphasis on 1961 and 1964, because of late summer drawdowns below what had been labeled critical pool levels. Based on the most recent operations study, this is no longer the case in Seminoe Reservoir. As was shown on figures 47 and 49, the d.f. for evaluating the probability of a late summer oxic hypolimnion was worse in 1965 than in the years 1961 and 1964 that were evaluated in the Pathfinder Reservoir temperature model. This is translated into probabilities of a late summer oxic hypolimnion in Seminoe Reservoir in Table 20.

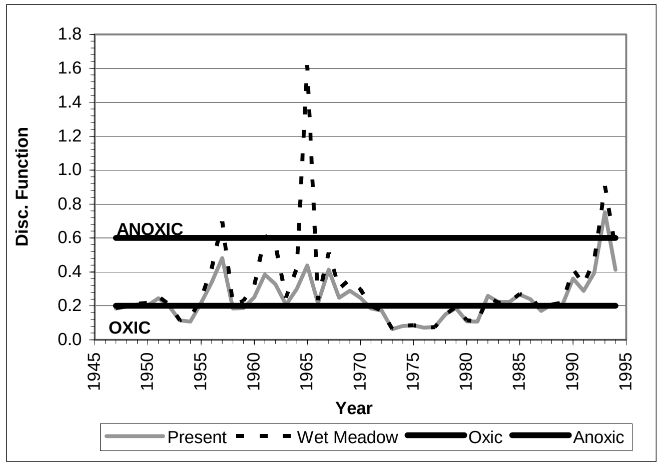


Figure 64. Annual discriminant functions for the Present Condition and the Wet Meadow Emphasis alternative in Seminoe Reservoir

Table 20. Probability of a late summer oxic hypolimnion in Seminoe Reservoir during the critical period: 1961-1965

Year	Present	Gov. Comm.	Water Emphasis	Water Leasing	Wet Meadow
1961	0.41	0.39	0.38	0.31	0.30
1962	0.49	0.49	0.48	0.45	0.35
1963	0.60	0.59	0.57	0.53	0.55
1964	0.48	0.43	0.42	0.34	0.40
1965	0.41	0.16	0.16	0.27	0.15

In the evaluation of the effects of drawdown on Lake McConaughy in the DEIS, the critical factor determining the temperature and DO conditions relative to trout habitat in the late summer was the April water level. The may be an effect seen here on the areal phosphorus loading. The pool levels in Seminoe in the spring of 1965 are low with a correspondingly small surface area and mean depth. The phosphorus load is relatively large in 1965, all of which leads to a large d.f. and a small probability of a late summer oxic hypolimnion for several of the alternatives (Table 20). In all of the measured profiles, the anoxic zone within the hypolimnion is confined to an area relatively near the bottom. In 1965, the pool level increased considerably because of a relatively large volume of runoff. There would likely be a relatively large hypolimnion under the circumstances shown in 1965, and trout habitat should not be greatly affected.

Figure 63 showed a much smaller d.f. than any of the other alternatives on the respective d.f. plots (figures 47, 49, and 51). The probabilities of a late summer oxic hypolimnion associated with those d.f. are shown in Table 20. The thing to note in Table 20 is the difference from the

Present Condition in 1961 and 1964 in the probabilities for the Full Water Leasing alternative, which are much lower. Although the Full Water Leasing alternative has a smaller maximum d.f. than the other alternatives, its d.f. is generally lower during the longer periods of drought. As a consequence, it shows something of a chronic effect which translates into the smallest overall probability in Table 19. As is indicated in tables 17 and 18, the least overall difference from the Present Condition, based on the probability of a late summer oxic hypolimnion, is shown by the Governance Committee alternative.

Oxic Conditions Comparison of Alternatives in Pathfinder Reservoir

Bottom DO measurements from Pathfinder Reservoir are shown on Figure 65. There are fewer DO profiles available for Pathfinder Reservoir than for Seminoe Reservoir. There were 34 measurements made over the course of 8 years at Pathfinder. As indicated on Figure 65, no measurements showed anoxia as defined by Reckhow and Chapra (1983). The minimum measured DO in the reservoir was 0.2 mg/L, which approaches anoxia, but does not meet the definition. This is not to say that anoxia has never occurred, just that it has not been observed. The discussion accompanying Figure 61 applies here as well. However, the probability that anoxia has been missed in the samples is somewhat greater in the case of Pathfinder Reservoir, because of the slightly smaller sample size and the shorter monitoring period, *i.e.* 8 years rather than 11. In the absence of any 0 mg/L DO readings, the historic frequency of anoxic conditions cannot be defined. However, because there was at least 1 very low DO measurement, anoxia should be assumed to have occurred at some time in the past nearly 100 years of the reservoir's existence.

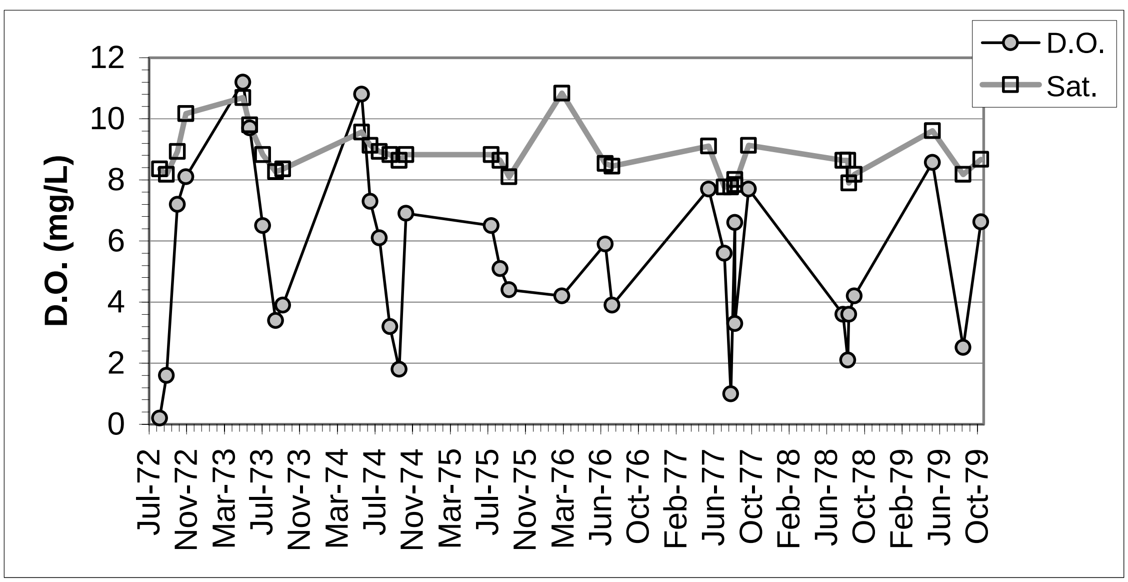


Figure 65. Bottom DO and saturation DO concentrations
in Pathfinder Reservoir between 1972 and 1979

Table 21 summarizes the distribution of the d.f. for Pathfinder Reservoir with respect to the oxic/indeterminate/anoxic probabilities of a late-summer oxic hypolimnion. For all of the

Table 21. Frequency (%) of discriminant function levels and mean probability of a late-summer oxic hypolimnion in Pathfinder Reservoir under the Present Condition and each of the action alternatives				
Condition/Alternative	% Oxic	% Indeterminate	% Anoxic	Probability oxic
	(d.f.[1] < 0.2)	(0.2 < d.f. <0.6)	(d.f. > 0.6)	
Present Condition	90	8	2	0.744
Governance Committee	85	13	2	0.710
Water Emphasis	81	17	2	0.698
Full Water Leasing	92	8	0	0.771
Wet Meadow Emphasis	88	12	0	0.694
[1] d.f. – discriminant function				

alternatives, the probability of a late-summer oxic hypolimnion is much higher than was the case for Seminoe Reservoir. The minimum percentage of d.f. with an oxic probability is 81 for the Water Emphasis alternative, while the greatest percentage is for the Full Water Leasing alternative. The percentage for the Full Water Leasing alternative is lightly higher than that of the Present Condition.

As was noted in the Methods section, the indeterminate category is probably representative of an oxic condition as well. If that is the case, then there is only a very small probability of a late-summer anoxic hypolimnion for the Present Condition and 2 of the action alternatives, Governance Committee alternative and the Water Emphasis alternative. The probability of a late-summer anoxic hypolimnion for the Full Water Leasing and the Wet Meadow Emphasis alternatives is virtually nil on the basis of the d.f.. The very low probability for the Present Condition would support the above discussion related to the observed minimum DO concentrations, which also indicate a very low probability of a late-summer anoxic hypolimnion, as defined by Reckhow and Chapra (1983). It may well be that anoxia has never occurred, or it may be that anoxia has not occurred since the construction of Seminoe Reservoir, which would serve as something of a nutrient trap and reduce the nutrient load to Pathfinder Reservoir.

The average probability of a late-summer oxic hypolimnion is shown in the last column of Table 21 for the Present Condition and each of the action alternatives. Where the equivalent probabilities were between 0.5 and 0.6 for Seminoe (Table 19), they are much higher for Pathfinder Reservoir, with most of the average probabilities around 0.7 and that for the Full Water Leasing alternative near 0.8 (Table 19).

Figure 66 shows a comparison between the d.f. of Pathfinder Reservoir for the Present Condition and the Governance Committee alternative. The time series of d.f. for the Governance Committee alternative Pathfinder Reservoir is very much like that of the one for Seminoe Reservoir that was previously shown on Figure 60. As was the case on Figure 60, the d.f. of the Governance Committee alternative overlays that of the Present Condition in most years, including the early 1990s, which is a departure from what occurred with the Seminoe d.f. The d.f. of the Governance Committee alternative is still much greater than that of the Present Condition during the period 1955 through 1970. The 1955 through 1970 period account for all of the increases in the d.f. that shift it from the oxic class of the Present Condition to the indeterminate, but probably oxic class of the Governance Committee alternative. Overall, 85 percent of the years have a d.f. in the oxic class (Table 19).

The one year that is consistent in the d.f. of the 2 reservoirs is 1965, when the d.f. of Pathfinder Reservoir is also near 1.6. The equivalent probability of a late-summer oxic hypolimnion in Pathfinder Reservoir is also 0.16 just as it was in Seminoe Reservoir in 1965 (Table 20). Such a probability would indicate that an actual late-summer anoxic hypolimnion would occur between 1 in 5 and 1 in 10 years under the conditions represented by 1965. But 1965 is only 1 year. The average probabilities of a late-summer oxic hypolimnion from the Governance Committee alternative and the Present Condition show little difference (Table 19), with that of the Governance Committee alternative about 0.03 lower than that of the Present Condition.

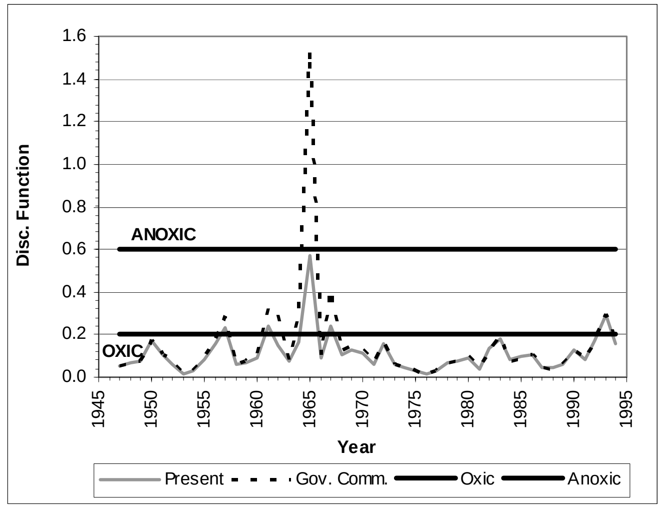


Figure 66. Annual discriminant functions for the Present Condition and the Governance Committee alternative in Pathfinder Reservoir

A comparison of the d.f. of the Water Emphasis alternative to that of the Present Condition appears on Figure 67. Once again the plot for Pathfinder Reservoir is much like the equivalent plot for Seminoe Reservoir. With the exception of 1965, the only year with a d.f. in the anoxic class, the annual d.f. in Pathfinder Reservoir for Water Emphasis alternative is as if the d.f. for the alternative in Seminoe reservoir were shifted downward toward the abscissa of the graph (compare figures 49 and 54). The Pathfinder Reservoir d.f. for 1965 is actually slightly higher than the one for Seminoe. As has been the case with many of the d.f. comparisons, the d.f. of the alternative is greater than that of the Present Condition in each of the years during the period 1955 through 1970, while in the remainder of the years, there is no apparent difference. The Water Emphasis alternative has the lowest percentage of d.f. in the oxic class, but there are still over 80 percent of the years in that class, and the highest percentage in the indeterminate, but probably oxic class of any of the alternatives, including the Present Condition (Table 19). Although the Water Emphasis alternative probability of a late summer oxic hypolimnion is below 0.7, it is little different from that of the Governance Committee alternative with a numerical difference of about 0.01.

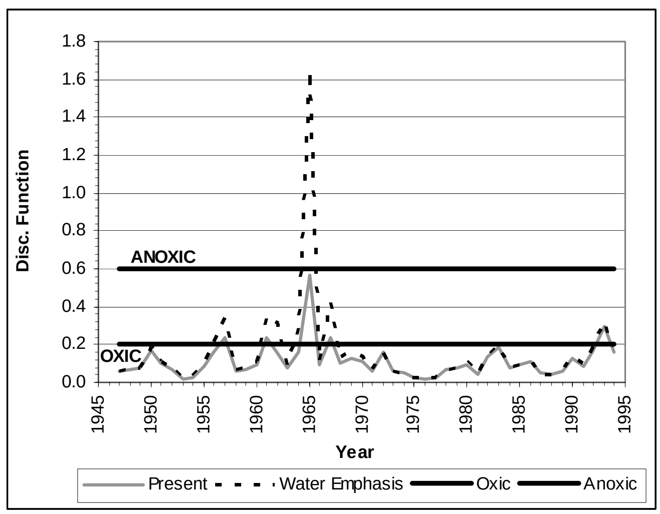


Figure 67. Annual discriminant functions for the Present Condition and the Water Emphasis alternative in Pathfinder Reservoir

In Pathfinder Reservoir the Full Water Leasing alternative shows a complete turnaround from what had been observed in Seminoe Reservoir in terms of the ranking of the probability of a late-summer oxic hypolimnion. In Seminoe Reservoir, the Full Water Leasing alternative ranked last with the lowest probability of any alternative (Table 19). In Pathfinder Reservoir, the Full Water Leasing alternative has the highest probability (Table 19). The Full Water Leasing alternative has no d.f. in the anoxic class and is tied with the Present Condition with the fewest percent d.f. in the indeterminate class (Table 19). As a result, the Full Water Leasing alternative has the highest percentage of d.f. in the oxic class. All of these factors are reflected in the high probability of a late-summer oxic hypolimnion.

Figure 68 shows a comparison between the time series distribution of the d.f. for the Present Condition and the Full Water Leasing alternative. In most year, as has been true of the other alternatives, the d.f. for the Present Condition and the Full Water Leasing alternative are overlain on the plot. However, unlike those of the other alternatives, the d.f. of the Full Water Leasing alternative are lower than those of the Present Condition in the more adverse periods of the record. For example, the d.f. for the Present Condition and the Full Water Leasing alternative are overlain during much of the 1955 through 1970 period, but the d.f. of the Full Water Leasing alternative is lower than those of the Present Condition in the years 1961 through 1965. The same is true of the early 1990s, when the d.f. of the other alternatives exceed those of the Present Condition. The high percentage of d.f. in the oxic class if also evident on Figure 68, as is the absence of any d.f. above the anoxic boundary. Overall, the Full Water Leasing alternative would appear to have a favorable effect on the probability of a late-summer oxic hypolimnion in Pathfinder Reservoir.

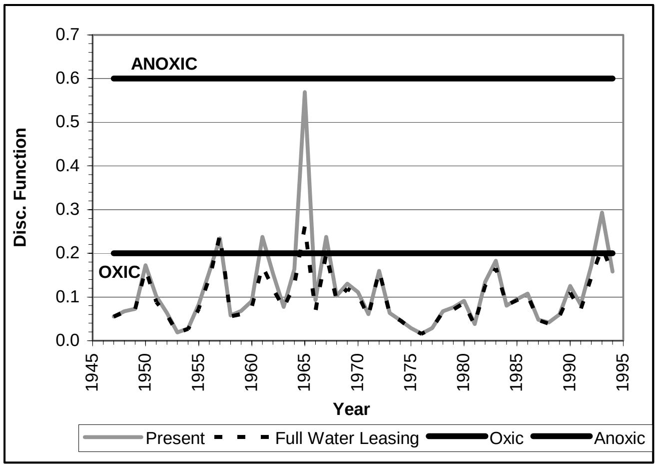


Figure 68. Annual discriminant functions for the Present Condition and the Full Water Leasing alternative in Pathfinder Reservoir

In Seminoe Reservoir, Wet Meadow Emphasis alternative showed a probability of a late-summer oxic hypolimnion very much like that of the Full Water Leasing alternative (Table 19). In Pathfinder Reservoir, the probabilities of a late-summer oxic hypolimnion between the 2 alternatives show the greatest difference between any 2 alternatives, including the Present Condition (Table 19). The Wet Meadow Emphasis alternative has the lowest probability of any alternative, including the Present Condition.

Figure 69 shows a time series plot of the d.f. of the Present Condition and the Wet Meadow Emphasis alternative. As was the case with the other alternatives, the temporal distribution of the d.f. in Pathfinder Reservoir is a downward shifted reproduction of the one in Seminoe Reservoir with the exception of that for 1965. The 1965 d.f. for the Wet Meadow Emphasis alternative in Pathfinder Reservoir is actually higher than the 1 for Seminoe Reservoir, although not greatly so. The d.f. of the Wet Meadow Emphasis alternative are noticeably greater than those of the Present Condition through the 1955 through 1970 period and the early 1990s. However, the d.f. of the Wet Meadow Emphasis alternative are also slightly lower than those of the Present Condition in the early 1970s, which is a somewhat wet period in the North Platte Basin. Nevertheless, the Wet Meadow Emphasis alternative would appear to be the alternative that would have the greatest effect on the probability of a late-summer oxic hypolimnion in Pathfinder Reservoir.

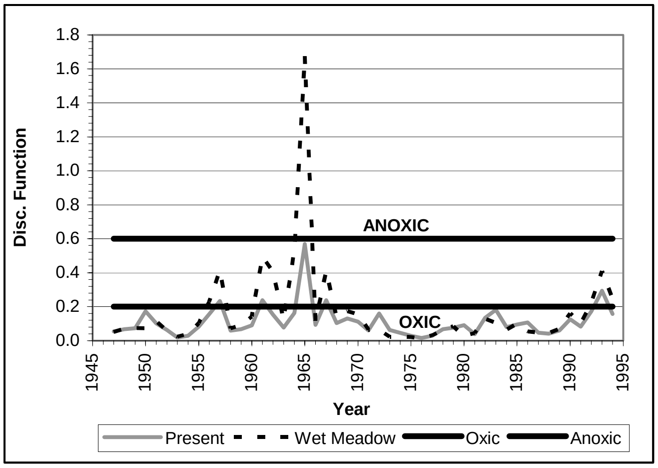


Figure 69. Annual discriminant functions for the Present Condition and the Wet Meadow Emphasis alternative in Pathfinder Reservoir

Since the whole purpose of this analysis is a result of concerns over the low water storage in the North Platte Reservoirs during the early 1960s, Table 22 presents the probability of a late-summer oxic hypolimnion in Pathfinder Reservoir for the Present Condition and each alternative for each year in that period. As has been shown above, the lowest probability of a late-summer oxic hypolimnion occurs in 1965 for each of the alternatives and the Present Condition. With the exception of the Full Water Leasing alternative, the probability of a late-summer oxic hypolimnion in 1965 in Pathfinder Reservoir for the action alternatives is about ½ that of the Present Condition and very much like the similar probabilities in Seminoe reservoir (refer to Table 20).

Table 22. Probability of a late summer oxic hypolimnion in Pathfinder Reservoir

Year	Present	Gov. Comm.	Water Emphasis	Water Leasing	Wet Meadow
1961	0.55	0.46	0.44	0.63	0.36
1962	0.66	0.51	0.49	0.72	0.42
1963	0.80	0.79	0.77	0.81	0.71
1964	0.64	0.50	0.50	0.70	0.35
1965	0.34	0.16	0.15	0.53	0.15

The Full Water Leasing alternative shows an increased probability of a late-summer oxic hypolimnion over the Present Condition in all of the years. None of the probabilities of a late-summer oxic hypolimnion associated with the Full Water Leasing alternative are below 0.5 (or 1 in 2), including 1965 (Table 22). Under the Present Condition, only 1965 has a probability of a late-summer oxic hypolimnion less than 0.5. The Governance Committee alternative has reduced probabilities of a late-summer oxic hypolimnion in Pathfinder Reservoir in all years relative to the Present Condition (Table 22), but less so than either the Water Emphasis or Wet Meadow Emphasis alternative. In summary, the Full Water Leasing alternative would increase the probability of a late-summer oxic hypolimnion in Pathfinder Reservoir, while the Governance Committee alternative would have the least adverse impact of those alternatives that have an adverse impact, *i.e.* all action alternatives except the Water Leasing Alternative.

Temperature and Dissolved Oxygen

The purpose of this section of the report is to pull together the results of the preceding 2 studies and develop a net impact of the various alternatives on the Pathfinder Reservoir fisheries and to relate the DO results to the temperature results. Because Pathfinder Reservoir would be subject to the most severe drawdown, it would be the most likely to be severely affected. Because the Governance Committee Alternative is the preferred alternative and because its results are similar to those of the Water Emphasis and Wet Meadow alternatives, the Governance Committee Alternative will be used as a surrogate for those alternatives. Since the Water Leasing Alternative generally shows an improvement over the Present Condition, its results are not a factor in this analysis.

The temperature study of Pathfinder Reservoir strictly related to the suitability of the reservoir to support trout during the late summer, when the warmest bottom conditions would prevail. If the reservoir became too warm for the trout, then the trout fishery would be lost and restocking would be necessary.

The previous section of the report is an attempt to evaluate the probability that the bottom layer of Pathfinder Reservoir would become anoxic. Even if the bottom temperature was within the temperature range that would support trout, the hypolimnion could become anoxic and become incapable of supporting trout or any fish for that matter. Theoretically, all of the DO that will be in the hypolimnion during the summer is what is present at the onset of stratification. As the summer progresses, DO is depleted, primarily by bacterial respiration in the sediments. Consequently, anoxia progresses from near the sediments to areas higher in the hypolimnion. Complete anoxia rarely occurs in the hypolimnion. There is diffusion of DO from the lower layers of the metalimnion into the upper part of the hypolimnion. However, this can also result in depletion of DO in the lower part of the metalimnion. Under these circumstances, part of the metalimnion may also have too little DO to support trout, which have a somewhat higher DO requirement than most fish.

Figure 41 indicates that the hypolimnion in early August with the Governance Committee Alternative is somewhat smaller than would be the case under the Present Condition, but it would still be relatively large. In both cases, the hypolimnetic temperature profiles are rather complex. However, by late August, the temperature profiles with both of the alternatives are more classic, but the hypolimnion with the Governance Committee Alternative is much smaller. Recall from Table 140 that the probability of an oxic late summer hypolimnion in Pathfinder Reservoir is less than 50 percent (0.46). Also recall that anoxia as defined by the results of the probability analysis means any measurement of 0 mg/L of DO anywhere in the profile. This does not really define how conditions would be in the hypolimnion. The anoxic hypolimnion may be a single measurement with much of the hypolimnion relatively oxic or it could represent a nearly anoxic hypolimnion throughout.

There have been no anoxic measurements in Pathfinder Reservoir. However, there was one occasion on which anoxia was observed in Seminoe Reservoir. Although there may be some question as to the applicability of the distribution of DO in Seminoe Reservoir to that of Pathfinder Reservoir, it seems worth exploring.

Figure 70 shows temperature and DO profiles in Seminoe Reservoir on August 15, 1977. Recall that 1977 was a drought year and the lowest reservoir conditions occurred in Pathfinder Reservoir in 1977. There is a decline in DO in the profile on Figure 59 that begins in the epilimnion, which extends to a depth of about 12 meters (39 feet). The decrease in DO begins immediately below the surface. The decrease continues to a depth of about 24 meters (79 feet). There is a further decrease in DO in the hypolimnion from just above 4 to 0 mg/L at the bottom. The thing of note is that the DO is less than 5 mg/L throughout the hypolimnion, to be exact, at all depths below 18 meters (59 feet). Alternatively, the maximum temperature within the epilimnion is 18°C (64°F) and within the tolerance of trout. Consequently, the actual trout habitat from a temperature perspective would extend throughout the reservoir, at least in its deeper areas. Near-shore temperatures tend to be warmer in the daytime than is the case in the deeper, more open areas of a reservoir, but much of the reservoir would be capable of supporting trout in the epilimnion.

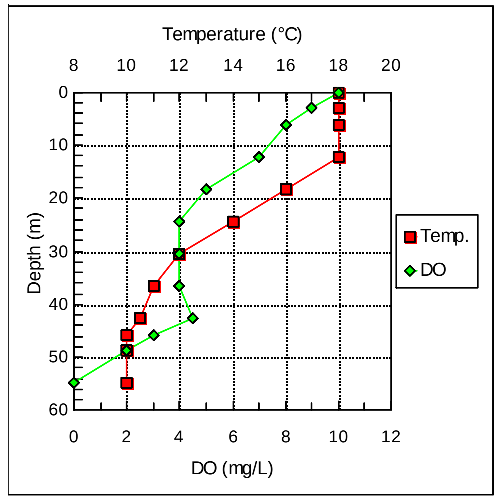


Figure 70. Temperature and DO profiles on August 15, 1977 in Seminoe Reservoir

During 1977, the minimum pool in Seminoe Reservoir was about twice that of Pathfinder Reservoir, *i.e.* 630,000 acre-feet *versus* 315,000 acre-feet at the time profiles were measured. Nevertheless, there was no anoxic measurement in Pathfinder Reservoir. However, there was very low DO (1 mg/L) at the bottom of the reservoir in mid-August 1977 and conditions were likely more severe for coldwater fish in Pathfinder Reservoir than they were in Seminoe Reservoir. The conditions in Pathfinder Reservoir on August 17, 1977, are illustrated on Figure 71.

There was only a small temperature change throughout the profile in Pathfinder Reservoir on August 17, 1977 (Figure 71). Much of the decrease occurred near the surface, which would indicate that the peak temperature at the surface is a reflection of surface heating. Below the surface, the temperature remains between 18 and 19°C (64 and 66°f) to a depth of 24 meters (79 feet), at which point the temperature began to decline to the minimum of 17.5°C (63.5°F) at the base of the profile at a depth of 38 meters (125 feet). There is a decrease in DO that coincides with this last decrease in temperature. However, the DO decrease is from 2 mg/L to 1 mg/L. Either DO concentration would be fatal to trout. From a depth of 8 meters (26 feet) and below, the DO was below 3 mg/L (Figure 71) on August 17, 1977.

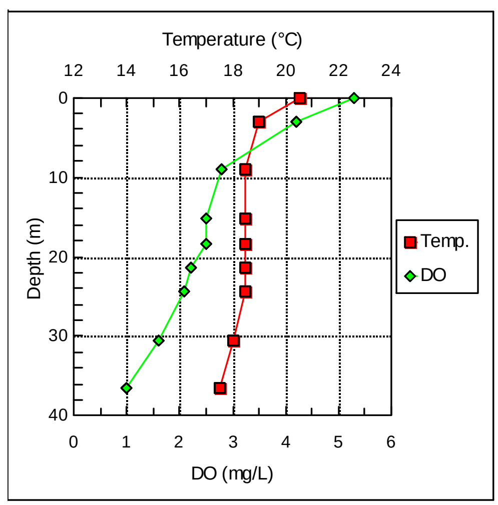


Figure 71. Temperature and DO profiles on August 17, 1977 in Pathfinder Reservoir

The maximum DO at the surface at the time was 5.3 mg/L. The DO at the next depth measured (3 meters or 10 feet) was below 5 mg/L. During mid-August 1977, the temperature and DO in Pathfinder Reservoir would have been on the upper edge for trout survival. From that perspective, anoxic conditions do not necessarily have to occur for conditions to be severe for trout in the reservoir. If there are records from 1977 in the Wyoming Game and Fish Department files on the conditions of the Pathfinder Reservoir trout fishery during the summer of 1977, these could probably provide a picture of the conditions that could prevail when there is a drawdown of Pathfinder Reservoir and the probability of anoxia is reasonably high.

The highest probabilities of anoxia in Pathfinder Reservoir occurred under the 1965 conditions. There was no temperature simulation performed for Pathfinder Reservoir for the 1965 conditions because the pool level in the late-summer was relatively high. To illustrate this, Figure 72 shows a plot of the 1964 and 1965 end-of-month contents of the reservoir in the 2 years. What happens under the 1965 simulated conditions is that the drawdown that occurs in the late summer of 1964 remains throughout the winter. Consequently, Pathfinder Reservoir remains at its minimum operating level throughout the winter. The method for estimating the probability of anoxia is based on areal phosphorus loading. Under the 1965 conditions, the overwinter pool is small. The maximum phosphorus loading occurs during spring runoff. In 1965, runoff was returning to a more normal level as reflected by the increased reservoir contents during the summer on Figure 72. However, the loading is being applied to a very small initial pool with a relatively small surface area. Consequently, the areal loading will be large and result in the large probability of anoxia later in the summer. How realistic is this?

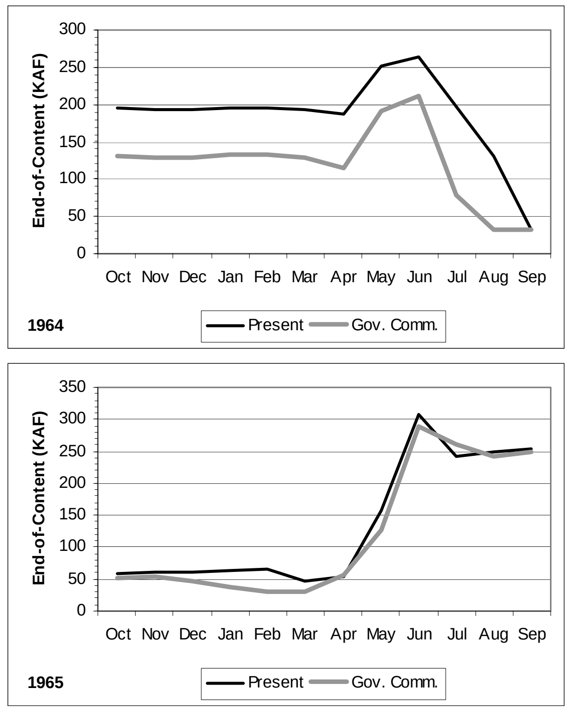


Figure 72. Present Condition and Governance Committee Alternative end-of-month content in Pathfinder Reservoir during 1964 and 1965

A large load applied to a small area or reservoir content will result in a relatively high concentration of phosphorus in the reservoir. The concentration of phosphorus in reservoirs is usually high during the spring under normal conditions because of the effect of spring runoff. The high spring phosphorus concentration typically fuels a spring phytoplankton bloom. Unlike the late summer phytoplankton bloom that occurs in the North Platte reservoirs consisting of the Cyanobacterium, *Aphanazomenon flos-aquae*, the spring bloom consists of diatoms. The spring bloom tends to be relatively brief. Diatoms have a test (shell-like outer layer) of silica and have no means of aiding flotation. As a consequence, when the reservoir quits mixing, diatoms settle to the bottom of the reservoir and help to fuel the depression of DO as they decompose. With the larger spring phosphorus concentration, a larger than usual spring bloom could occur and lead to a larger than usual depression of DO. This is the underlying theory for the DO probability estimation. On this basis, the results of the estimates for the probability of an late-summer anoxic hypolimnion may not be unrealistic. On the other hand there would likely be a larger hypolimnion to work with and an increased probability of some area of suitable temperature and DO in the reservoir.

One further consideration in relation to trout habitat in Pathfinder Reservoir (and Seminoe Reservoir as well) can be illustrated on the basis of the data presented earlier on Figure 28. The plot shows the inflow temperatures of the North Platte and Sweetwater rivers upstream from Pathfinder Reservoir. The North Platte River at that point is the release from Seminoe Reservoir as it is warmed slightly in the reach between Seminoe Dam and the gage site, while the Sweetwater River is essentially free-flowing and reflects ambient conditions only. The peak temperature in the North Platte River at that site occurs in August or September on the average, depending mostly on the time of reservoir mixing following the break down of stratification. Alternatively, the Sweetwater River shows its peak temperature in July on the average. The decrease of the temperature of the Sweetwater River in August reflects the normal temperature regime in relation to ambient conditions. This temperature regime is also what could be expected of the epilimnion of a reservoir. On the average, the temperature of the Sweetwater River drops below 20°C in August. This is consistent with the results of the temperature simulation of Pathfinder Reservoir. For this reason, if the hypolimnion becomes anoxic, there is likely to be a refuge for trout in the epilimnion. However, this may not always be the case. On Figure 28, the error bars about the August average temperature are much larger than those for July. This would indicate that temperatures as high as July can occur in August, most likely during drought conditions when low flows allow greater warming. Nevertheless, the above described mechanism may apply in some years.

North Platte River, Nebraska

The North Platte River in Nebraska would also be affected by the delivery of water to the Middle Platte River from the Wyoming mainstem reservoirs. The North Platte River in the vicinity of the Nebraska part of the North Platte Project is shown on Figure 73. Like the river in Wyoming, the primary indicator used to evaluate the effects of the Program alternatives on water quality is TDS. The analysis of the effects of Program alternatives on TDS is presented in this section of the water quality appendix. However, also like the river in Wyoming, the North Platte River in Nebraska is listed as impaired, particularly because of excessive bacterial concentrations during the summer recreation season, but unlike the situation in Wyoming, a TMDL has been prepared for bacteria (fecal coliform bacteria) in the North Platte River between the State Line and Lake McConaughy. The bacterial standard only applies during the summer when body contact recreation is likely to occur. Although background on bacteria is presented in this section of the appendix, the impact analysis appears in the later section on the effects of the Program alternatives on waters listed as impaired under Section 303(d) of the Clean Water Act.

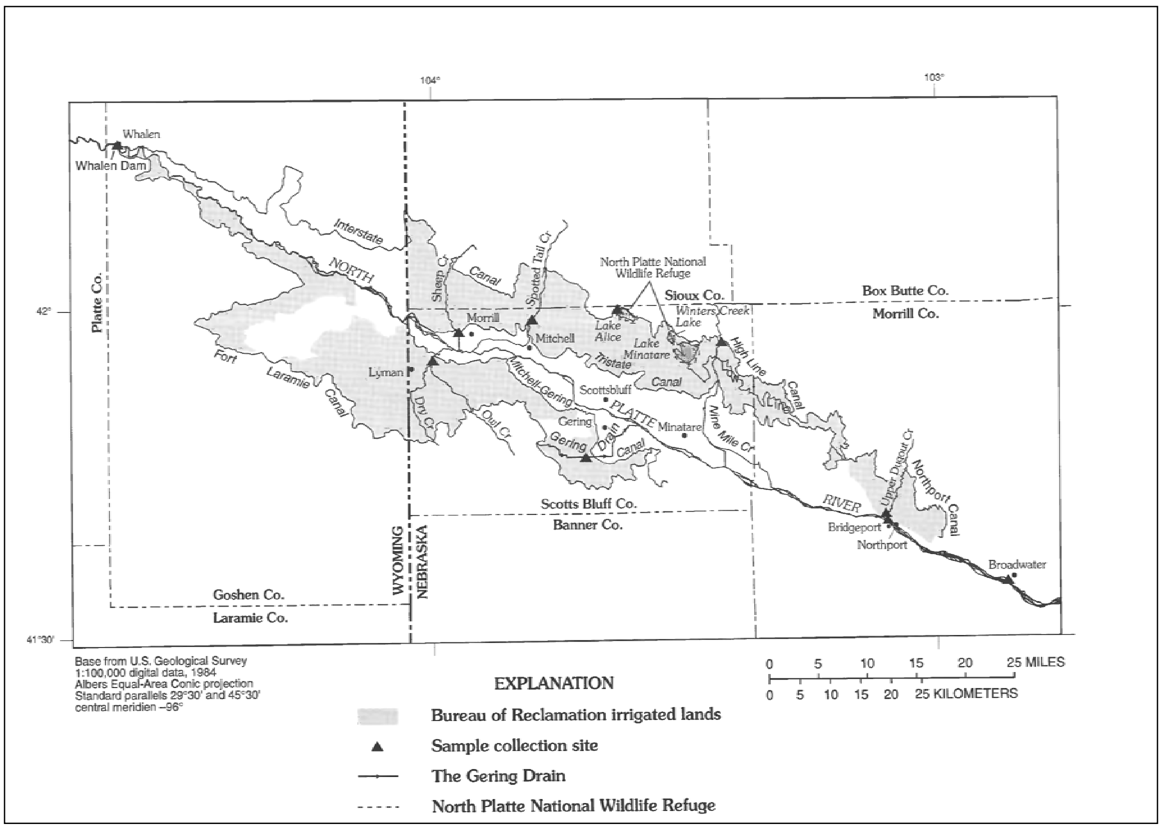


Figure 73. North Platte Project lands between Whalen Dam, WY, and Broadwater, NE
(Source: Druliner *et al.*, 1999)

During the review of the DEIS, concerns were raised over the potential effects of the Program on coldwater fisheries in the North Platte River and several of its tributaries in Nebraska. Water quality related to conditions for these fisheries will also be reviewed in this section of the appendix.

TDS of the North Platte River, Nebraska

Water quality data were retrieved from the USGS NWIS website for the North Platte River basin in Nebraska. The TDS data for mainstem sites are summarized in Table 23. The minimum and maximum at the sites upstream from Lake McConaughy each shows an increase from the State Line gage to Lisco. However, the median shows a peak at McGrew, the intermediate site. The number of samples is about the same at the State Line and Lisco sites, but there are fewer samples at McGrew. Although the trend of increasing TDS in the downstream direction as shown be the minimum and maximum TDS concentrations is what is expected, the absence of such a trend for the medians is not. The lack of a trend appears to be due to the difference in the periods of record at the 3 sites.

Table 23. TDS (mg/L) of the North Platte River from the State Line to the Keystone Diversion

Site	Minimum	Median	Maximum	No. of Obs.
WY-NE State Line	378	573	695	145
McGrew	444	644	715	105
Lisco	451	613	744	149
Keystone	477	501	573	25

There is a decrease in the minimum, median, and maximum TDS between the Lisco and Keystone sites on the North Platte River, despite the fact that there are far fewer samples at the Keystone site. The decrease in TDS between the 2 sites reflects the effects of the mixing of the more dilute and concentrated water in Lake McConaughy, as was described above for the North Platte Basin mainstem reservoirs in Wyoming.

The North Platte Basin operations model also includes the North Platte River in Nebraska. The North Platte Project diverts a large percentage of the stored water at Whalen Dam to canals that deliver water to Nebraska, including water to be stored in the Inland Lakes near Minatare. The Lewellen gage is the only mainstem site included in the operations model. The Lewellen gage is not a USGS water quality monitoring site, although the Nebraska Department of Environmental Quality samples the site as part of their monitoring program. Some of these data will be reported later in this section of the appendix. However, of the sites included in Table 23, Lisco is the nearest to the Lewellen gage. Because of the limited data available for Lewellen, Lisco data will be used as a surrogate.

Figure 74 shows the available TDS data from the Lisco site on the North Platte River. Figure 74 shows two things of note. The first relates to the intensity of sampling over time. The sampling frequency in the early part of the period of record is much more intensive than that of the more recent period. In the 1970s, samples during the irrigation season were collected weekly. The frequency decreased to monthly, then to bimonthly, and finally to quarterly at the end of the record. The second thing of note is the relatively small range in TDS in most years. The long-term average (600 mg/L) TDS is also shown on the plot. The tight grouping is illustrated by the standard deviation of 50 mg/L. For this reason, there appears to be little response of TDS to changes in flow. If this is the case, then little or no change would be expected due to flow changes related to the Program alternatives. The following explores this relationship (or lack of one) in more detail.

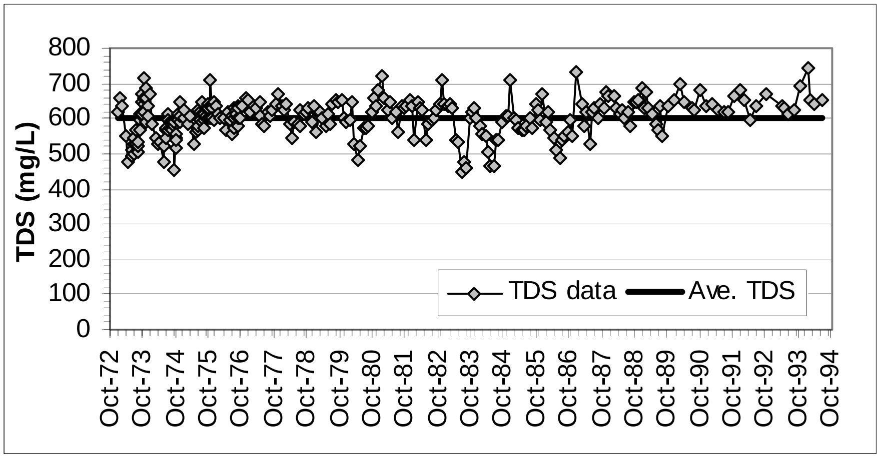


Figure 74. Time series plot of TDS data from the USGS Lisco gage

As is the case at most USGS monitoring sites, there are 2 measures of TDS at the Lisco gage. There is a gravimetrically measured TDS (residue on evaporation or ROE) and a TDS calculated from the sum of the major cations and anions. Because there were samples with the major ion data available, but no calculated TDS, additional TDS data were calculated on those dates; these calculated were used to supplement the USGS calculated TDS data. Each of these data sets was regressed on EC data. In addition, a TDS data set was created using the calculated TDS supplemented with ROE data. The TDS on Figure 74 represent the last set of TDS data supplemented with data calculated from the EC regression based on the last of the above described TDS data sets. This increased the number of TDS data points from the 145 shown in Table 23 to 300 for inclusion in the plot on Figure 74.

The relationship between the TDS data set shown on Figure 74 and associated flows is shown on Figure 75. As can be seen by the r^2 of the regression associated with the scattergram on Figure 75, the relationship is rather poor. The r^2 is lower than any of the regressions shown above in Table WY2 above, except for those of sites below dams. Although there is no dam immediately upstream from the Lisco site, the river is nevertheless highly regulated. The flow regulation above the site is apparently sufficient to negate the natural relationship between TDS and flow at the site. As can be seen on Figure 75, the small range in TDS (confined to less than 1 order of magnitude total range) is spread over a rather wide range of flow.

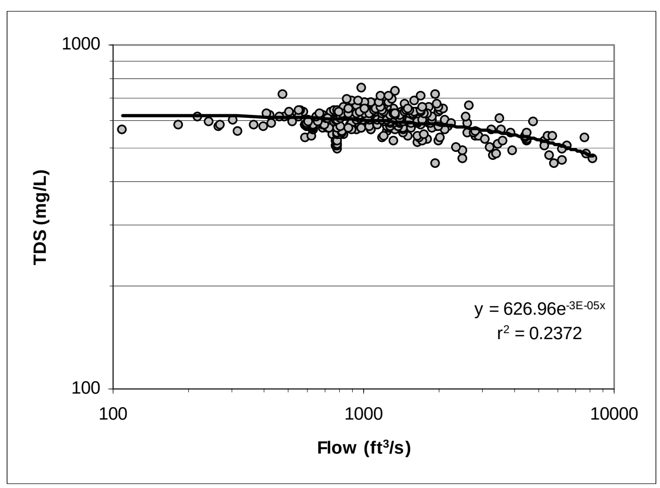


Figure 75. Relationship of TDS to flow at the Lisco gage

Sources of flow in the North Platte River reach consist of bypasses at the Whalen Diversion Dam and local inflows consisting of storm runoff, irrigation return flows, and discharge of ground water from the Sandhills aquifer. These sources vary seasonally. Because of this variation, seasonal relationships were also explored. The best of these are shown on Figure 76. The irrigation season TDS-flow data on Figure 76 contain all of the high and low flow data on Figure 75. The irrigation season regression equation on Figure 76 is also similar to that on Figure 75. However, the removal of the nonirrigation season data improves the r^2 of the regression considerably, more than doubling it to 0.5 on Figure 76.

The data for the nonirrigation season show a limited range of both TDS and flow (Figure 76). The form of the nonirrigation season regression differs from the one for the irrigation season. The r^2 of the nonirrigation season regression is similar to that of the overall regression on Figure 75. Because of the overall improvement in the fit of the irrigation season regression, the overall relationship is an improvement over the TDS-flow regression on Figure 75.

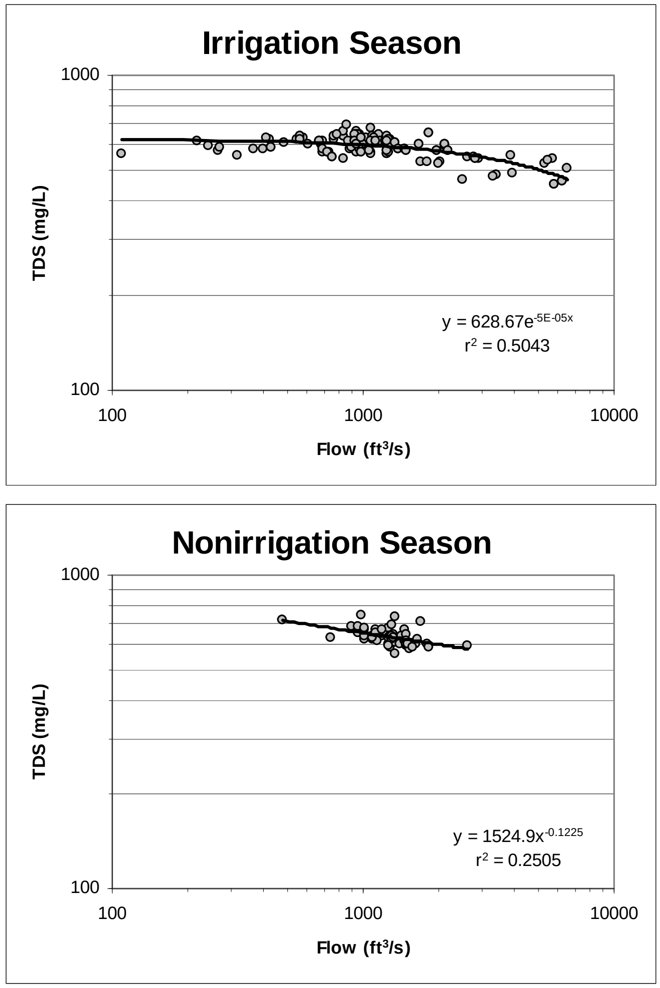


Figure 76. Seasonal regressions of TDS on flow at the Lisco gage

Bacteria in the North Platte River, Nebraska

The USGS monitored fecal coliform bacteria at the Lisco gage beginning with Water Year 1977. The bacterial data for the Lisco gage were retrieved from the USGS NWIS database. The complete fecal coliform data set is plotted on Figure 77. The plot also shows the Recreation Use Standard for fecal coliform bacteria. The line defining the standard is dashed because it only applies during the summer when body contact recreation is most likely to occur. The length of the dashes

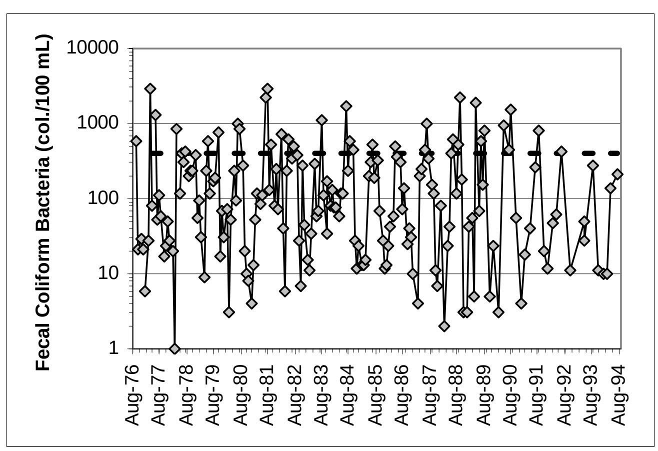


Figure 77. USGS fecal coliform data from the Lisco gage

on Figure 77 are proportional to the recreation season as defined by the fecal coliform database. The space between the dashes is the period when the fecal coliform standard does not apply.

There are numerous fecal coliform samples that plot above the standard on Figure 77, although the most recent data do not. The States in their evaluations of impaired water use only very recent data. Even the most recent samples on Figure 77 were collected more than 10 years ago. The States usually base their assessments on the most recent 5-year period, if possible.

The most recent bacterial samples on the North Platte River in Nebraska were collected in 2004 by the Nebraska Department of Environmental Quality (NDEQ). Those samples were analyzed for *Escherichia coli*, rather than fecal coliforms. *E. coli* is a fecal coliform. In 2001, samples were collected by NDEQ for both *E. coli* and fecal coliforms (as well as Enterococci) from the North Platte River at the Lewellen gage. The relationship between *E. coli* and fecal coliform bacteria based on the 2001 *E. coli* and fecal coliform samples is shown on Figure 78. *E. coli* may make up the majority of the fecal coliforms. Figure 78 would indicate that this was the case in the 2001 NDEQ samples.

Bacterial concentrations are highly variable. One of the EPA criteria for fecal coliforms is based on a geometric mean. For this reason, the bacterial data on Figure 78 are plotted on a log-log scale. The slope of the regression line is 0.88, which indicate that on the average, the log of the *E. coli* concentration is 88 percent of the log of the fecal coliform concentration. To determine the true relationship, the log-log ratio needs to be converted to actual concentrations. This can be done using the ratio of the geometric means of the two bacterial concentrations, which is 79 percent.

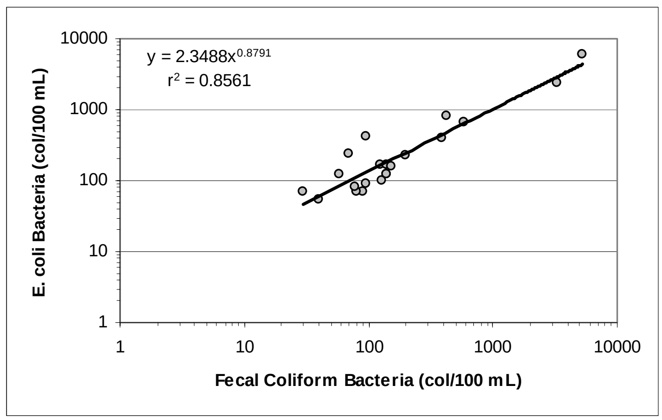


Figure 78. *Escherichia coli* and fecal coliform concentrations in the North Platte River at Lewellen during the Summer 2001 (Source: NDEQ)

The 2004 NDEQ *E. coli* concentrations are shown on Figure 79. The fecal coliform standard is also shown on Figure 79. As can be seen, the standard is exceeded periodically at all of the sites. The most obvious time at which the standard is exceeded is during the week of September 6, when all but one of the sites had *E. coli* concentrations above the standard. The only site at which the *E. coli* concentration was below the standard was at North Platte. The standard was exceeded least often at the two most upstream sites, the Stateline and near Morrill. The standard was exceeded twice at North Platte, but one of those times was outside of the of the period at which the other sites were sampled (Figure 79).

The main effect of the Program is to increase flow in the North Platte River during the delivery of water to Lake McConaughy. Ordinarily, if there is a relationship between bacterial concentrations and flow, it is positive. If this is the case, then an increase in flow would have the potential to increase bacterial concentrations in the river.

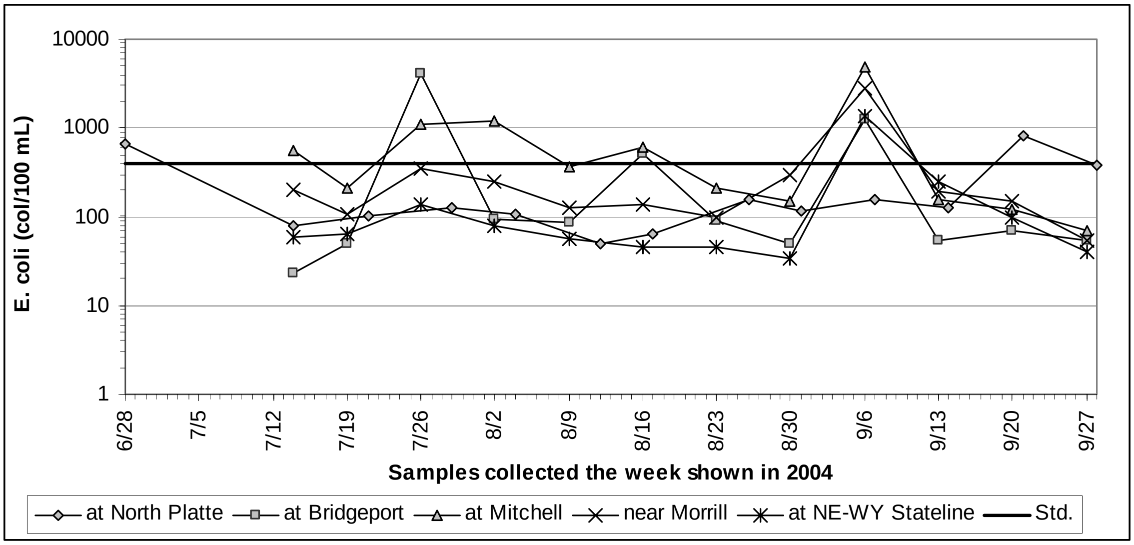


Figure 79. *Escherichia coli* concentrations in the North Platte River during the summer 2004

Figure 80 shows a scattergram of the long-term fecal coliform data from the Lisco gage plotted against flow. Figure 80 also shows a plotted trendline for the data. To some extent, the data distribution resembles that of the TDS-flow relationship at Lisco that was shown on Figure 75 in that the bacterial data are centered on the flow data. However, in this case, the r^2 of the trendline indicates there is no relationship between fecal coliform concentrations and flow. Based on the Lisco fecal coliform data, changes in flow would not be likely to affect fecal coliform concentrations in the North Platte River.

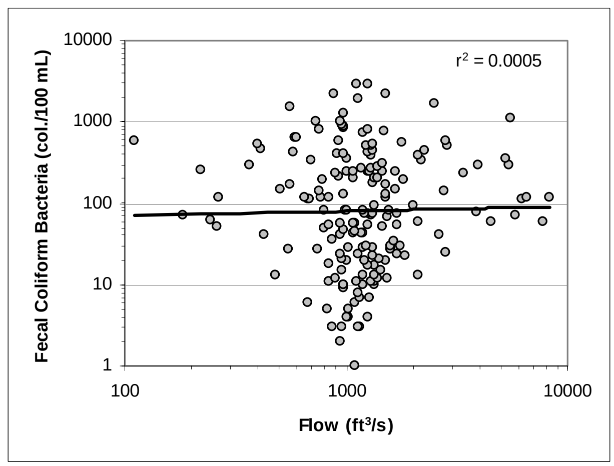


Figure 80. Relationship between fecal coliform concentration and flow at the Lisco gage

Temperature and Dissolved Oxygen in the North Platte River

During the review of the DEIS, there were a number of comments related to the fishery in the North Platte River between the State Line and Lake McConaughy. The effects were predicated on changes in flow that would affect the temperature and/or dissolved oxygen concentration of the river. Particular concern was raised over the affects of water leasing, which has the potential to remove water from irrigated lands along the river in Nebraska. The point being made is that irrigation augments the release of cool ground water from return flows. Another point relates to depletions of flow, but this is not directly related to water quality. However, as will be discussed in the Central Platte section, there may be an indirect relationship between flow and temperature.

USGS Monitoring Data

The long-term temperature record for the North Platte River at Lisco is shown on Figure 81. The temperature measurements are made weekly during the summer during the first several years of the record and monthly in all months beginning in 1977. Figure 81 also shows a temperature criterion for coldwater fish. The criterion is 20°C (68°F). As can be seen from Figure 81, that criterion is exceeded in most years. This fact would indicate that the river is at best marginal for coldwater fish during most summers.

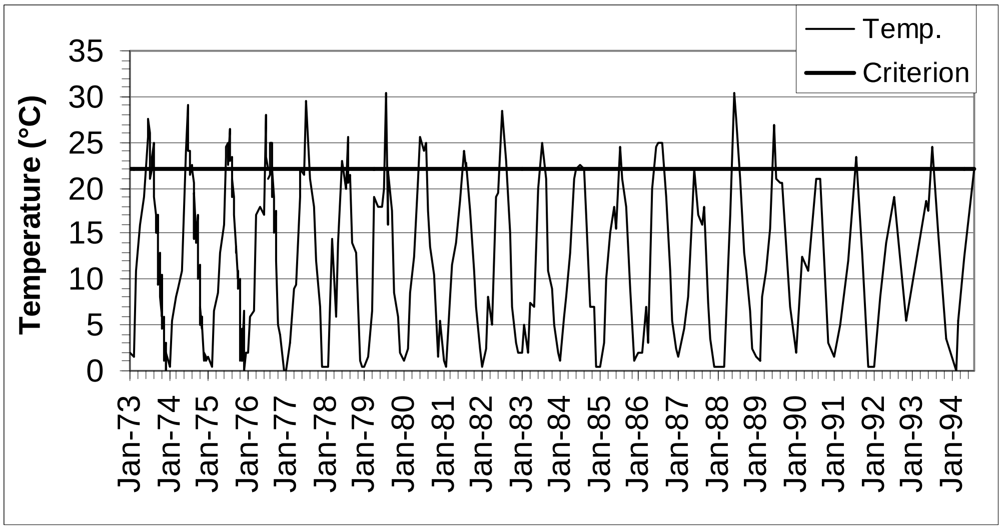


Figure 81. USGS Monthly temperatures at Lisco from 1973 through 1994

DO measurements were made at the same time as the temperature measurements. The DO requirement is generally higher for coldwater fish than it is for warmwater fish. Oxygen is more soluble in cold water than it is in warm water. The solubility of oxygen as a function of temperature is shown on Figure 82. The concentration of DO at saturation decreases from just below 15 mg/L at a temperature near freezing to about 7 mg/L when the temperature is at 30°C (86°F). This indicates that there should be less DO available for warmwater fish when the DO is at saturation.

The long-term DO concentrations and the percent saturation at Lisco are shown on Figure 83. Based on Figure 82, the DO concentration should be much higher in the winter than in the summer. However, in the absence of oxygen consuming factors, the saturation should remain near 100 percent.

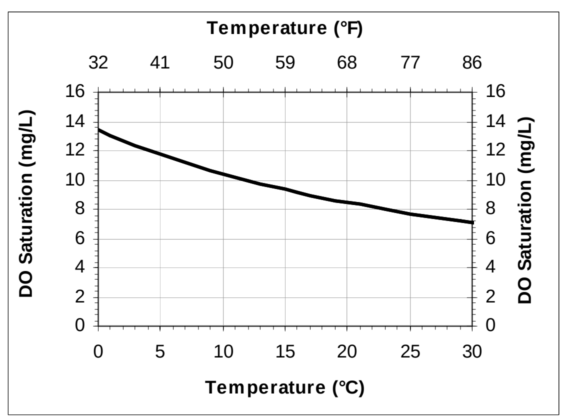


Figure 82. DO saturation concentrations as a function of water temperature

Early in the record, the DO and the percent saturation show considerably more variation than later in the record (Figure 83). This is likely due to the more frequent sampling in the first several years, but is also likely due to other factors during the period from the late-1070s through the early 1980s. Since the mid-1990s, the DO has shown the expected seasonal variation, but

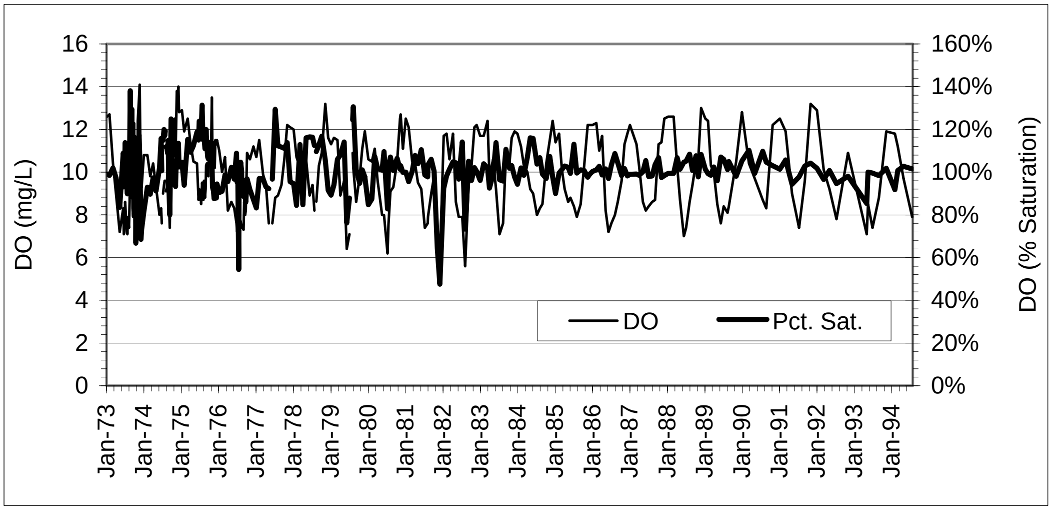


Figure 83. USGS Monthly DO at Lisco from 1973 through 1994

remained near saturation. The exception is near the end of the period of record in 1993, when the DO dropped below 80 percent of saturation. In general, coldwater fish can tolerate DO as low as about 75 percent saturation without undergoing excessive stress. On this basis, the DO should be adequate for support of coldwater fish.

The measurements shown on figures 81 and 83 were all made during the day. Just as there are seasonal variations in both temperature and DO, there are also diurnal variations in both. For several years, the USGS measured diurnal temperature and DO distributions once a year at the Lisco gage. Those data are plotted on Figure 84. The measurements were made in late August in 3 of the 4 years during 1978 through 1981; the measurements were made in early August in 1980. In 1978 and 1981, the data were reported every 2 hours, while in 1979 and 1980, the data were reported hourly.

In the August 1978 data, the water temperature at the Lisco gage peaked at 4 in the afternoon (16:00) and declined to its daily minimum at 6 in the morning (Figure 84). This is consistent with the generally expected diurnal distribution of water temperature. DO shows its peak at 6 AM, when the temperature is at its minimum. Although this is consistent with the relationship shown on Figure 82, the DO concentration is greater than the saturation concentration. This result is not what would be expected in nature. In the absence of photosynthesis, plants (algae) respire, just like animals do. Because of this, DO usually decreases during the night. The increase in DO in the morning of August 25, 1978, coincided with an increase in flow. This may indicate that there was an overnight shower. Rainwater tends to become oversaturated with DO as it fall through the atmosphere. An influx of rainwater could increase the DO above saturation.

The diurnal temperature distributions in the remaining 3 years plotted on Figure 84 all fit the expected patterns. Peak temperatures occurred in the late afternoon. In each of the 3 years, the peak daily temperature was between 25 and 30°C (77-86°F), although the peak in 1979 was just at 25°. The maximum daily temperature in the 4 sampled years on Figure 84 occurred in 1980. That peak temperature occurred continuously from 3 (15:00) to 6 (18:00) in the afternoon.

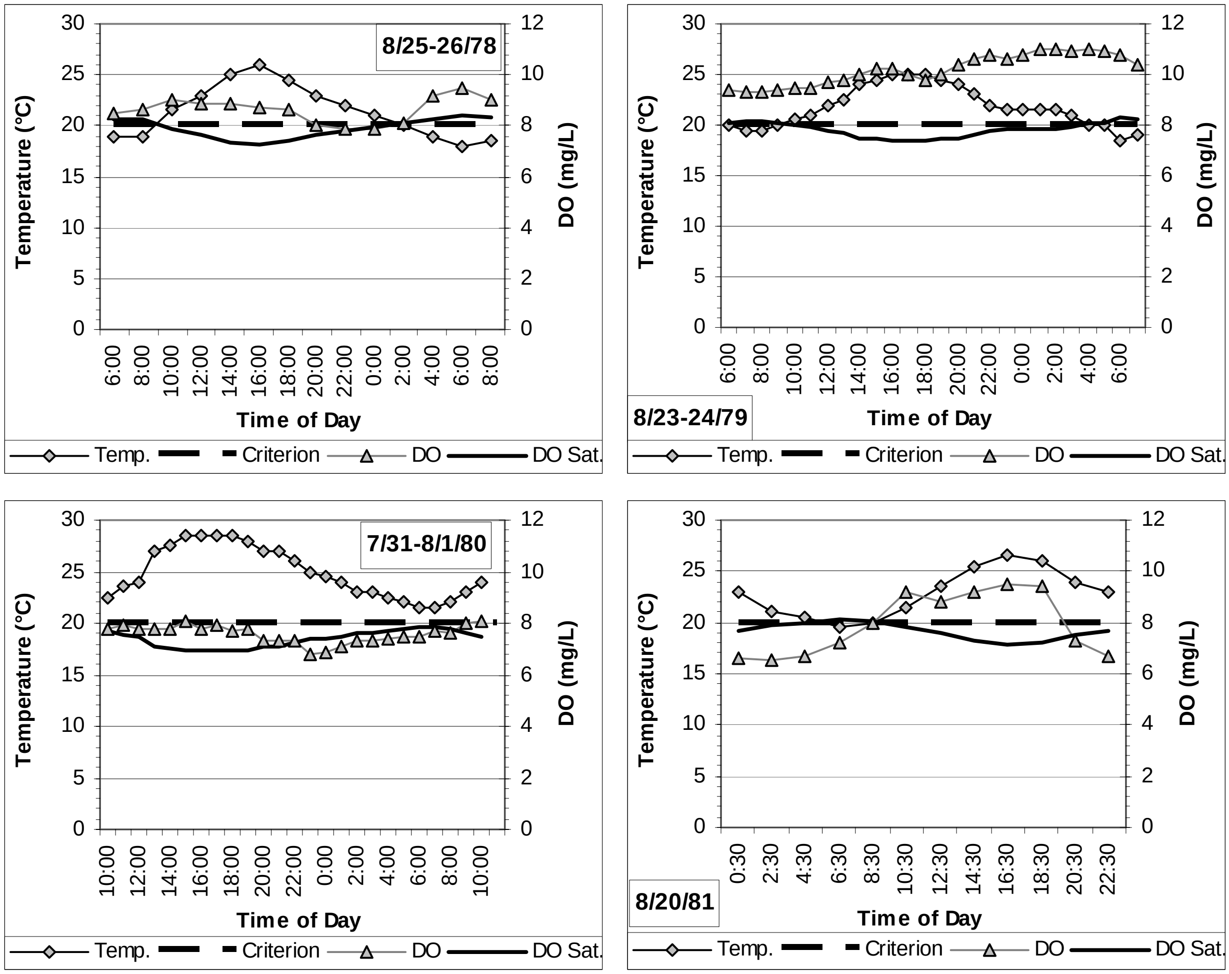


Figure 84. Diurnal temperature and dissolved oxygen of the North Platte River at Lisco during 4 summer days

The DO in 1979 remained well above saturation in all of the measurements on August 23 and 24 (Figure 84). As was the case in 1978, the peak DO occurred during the night, and all of the night time measurements exceeded the daytime measurements. However, there was no indication of an increase in flow during 1979, as had occurred in 1978. The flow measurements in 1979 were made in time increments of between 3 and 6 hours, and small peaks could have been missed. Nevertheless, the measurements that were made showed a decrease in flow throughout the night. There is no good explanation for the relatively high night time DO, other than the data may be questionable.

The diurnal patterns of DO in 1980 and 1981 are consistent with what is normally expected. In both years, the DO rises above saturation during the day when photosynthesis should exceed respiration and falls below saturation at night when respiration should be all that occurs. The deviations between the maximum and minimum DO and the equivalent saturation concentrations were much greater in 1981 than they were in 1980 (Figure 84). In 1980, the deviations from saturation are less than 1 mg/L during the night and most of the day (maximum difference of 1.2 mg/L at 3 PM), while in 1980, the DO was more than 1 mg/L below saturation during most of the night and exceeded saturation by over 2 mg/L during the afternoon. This result would indicate a much greater degree of biological activity during 1981, than in 1980.

Nebraska Department of Environmental Quality Data

The NDEQ measured temperature and DO at a number of sites on the North Platte River between 2001 and 2003. The temperature and DO measurements were made weekly during the summer of 2001 at 9 sites on the mainstem of the North Platte River between the State Line and North Platte. During 2002 and 2003, the measurements were made monthly at 4 sites. The 2002 measurements were made in all 12 months at 2 sites (Lewellen and North Platte), but did not begin until June at the remaining 2 sites (the State Line and Bridgeport). Measurements were made year-round at all 4 sites in 2003.

The NDEQ temperature data from 2002 and 2003 are shown on Figure 85. The North Platte River is classified as Coldwater Class B from the State Line to the mouth of Scout Creek. Of the sites on Figure 85, this includes all of the sites except for the one at North Platte, which is classified as Warmwater Class A. The distinction between the class A and B aquatic life categories relates to the support of reproduction. The temperature criterion for Coldwater Aquatic Life (22°C or 72°F) is shown on Figure 85. The temperature criterion for Warmwater Aquatic Life is 90°F (32°C), which is not shown, in that it is much greater than any of the measured temperatures shown on Figure 85. Alternatively, most of the summer temperatures exceed the 22°C criterion at all sites on Figure 85.

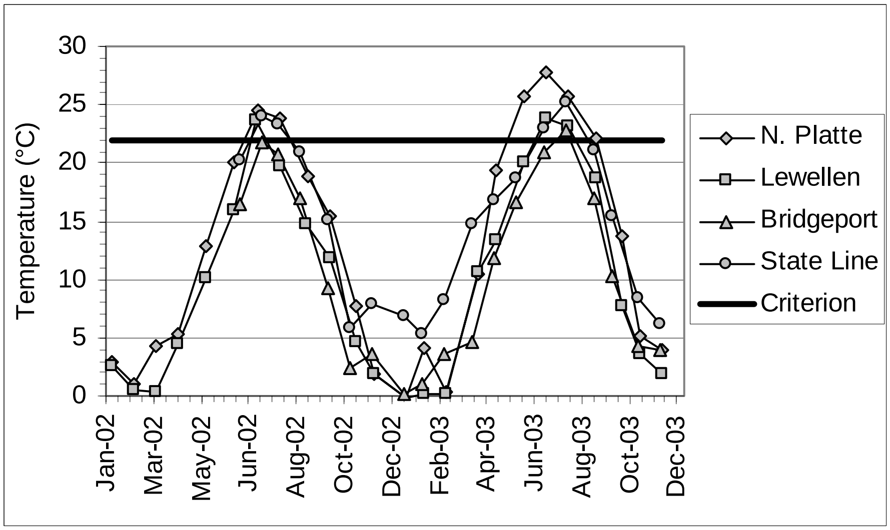


Figure 85. Monthly temperatures at 4 sites on the North Platte River in 2002 and 2003

Figure 86 shows the NDEQ weekly summer temperature data from 2001. The plots are arranged from downstream to upstream, in accordance with the NDEQ numbering system. Once again, all of the sites are classified for Coldwater Aquatic Life, except for the one at North Platte. The first 4 sites on Figure 86 are located downstream from Lake McConaughy.

The North Platte sites near Sutherland and Keystone are generally below the coldwater aquatic life criterion of 22°C throughout the summer. As was illustrated above, the temperature varies during the day and usually peaks late in the afternoon. The measurements at the Sutherland site were usually made in the morning, while those at Keystone were made around noon. On this basis, the temperature measurements at the 2 sites would not represent the maximum daily. Still, the 2 sites are the nearest to Lake McConaughy, which may exert some control on the temperatures of the sites. This would be case when the flows represent Lake McConaughy releases. If all or most of the flow is diverted, then the water temperatures would represent the

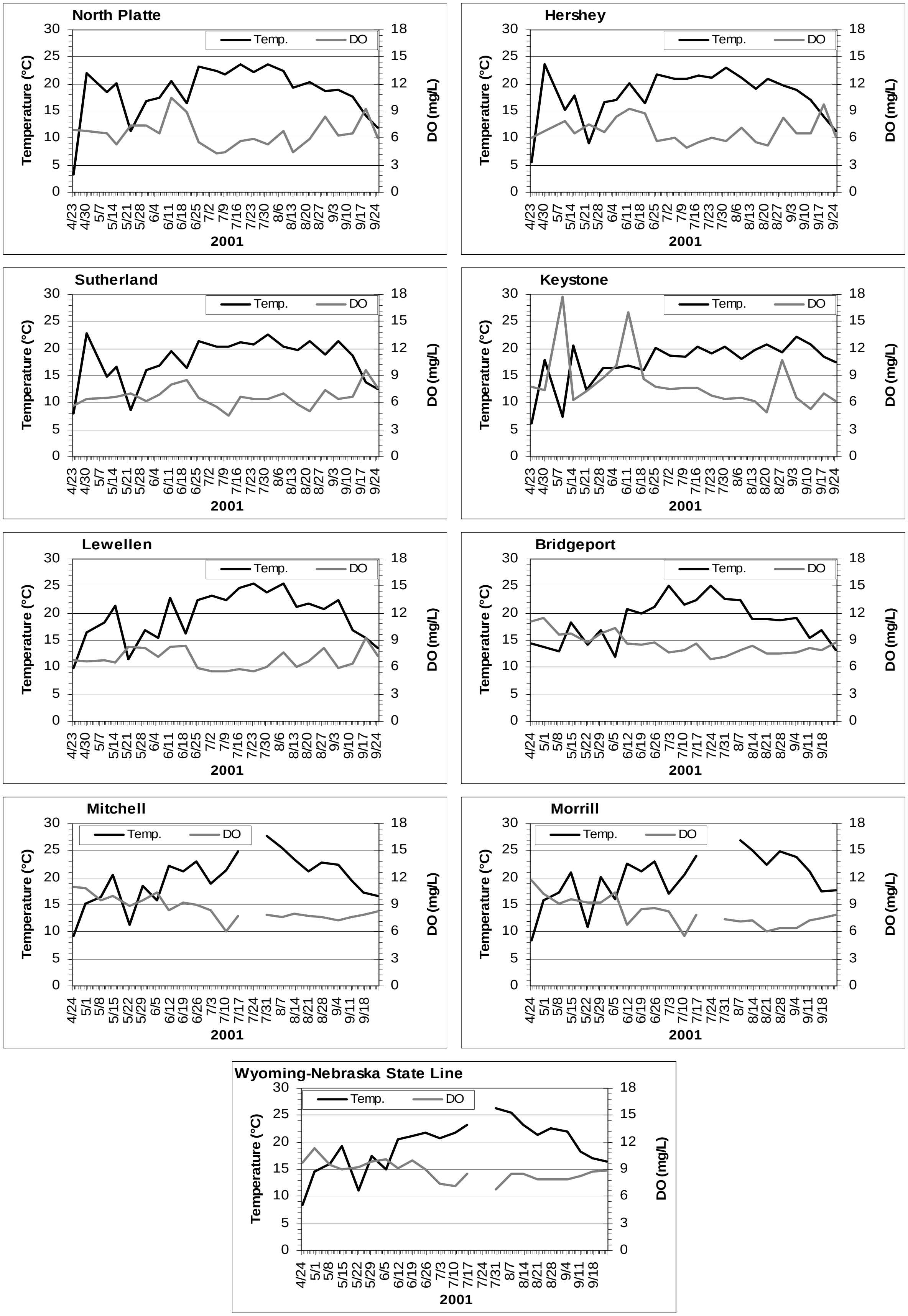


Figure 86. Temperature and dissolved oxygen at 9 sites on the North Platte River in Nebraska during 2001

temperature of the gains in flow above the sites. In the case of the Keystone data, the temperature is mostly below 20°C (68°F); the exception is the first 2 weeks in September, when the temperature was slightly above the standard (22°C) on one date (Figure 86).The first site upstream from Lake McConaughy is the site near Lewellen. The temperature at the Lewellen is generally around 25°C (77°F) during most of July and the early part of August (Figure 86). During that part of 2001, the temperatures at Lewellen were the overall highest of any of the sites. However, measurements for part of the period are missing at 3 of the sites farther upstream. The first measurement following the gap in the data at each of those sites is the highest of the year and exceeds 25° at each of the sites. In the case of the site at Mitchell, the temperature was several degrees greater than 25° (Figure 86).

Figure 86 also shows the DO concentration at each of the sites during the summer of 2001. The water quality criteria for DO that are associated with the coldwater aquatic life use classification are a bit more involved than the criterion for temperature. The DO criteria are for the most part based on mean minimum concentrations, although there is a 1-day minimum of 4 mg/L as well (NDEQ, 2002a). Because the measurements are all made during the daylight hours, it is unlikely that the daily minimum DO is represented by any of the data on Figure 86. The 7-day mean minimum DO criterion is 5 mg/L, while the 30-day mean DO criterion is 6.5 mg/L. In general, the DO criteria apply during the period of July 1 through March 31 or ¾ of the year. The remaining 3 months coincide with spring runoff, when the flows in the river tend to be cold and turbulent and low DO is unlikely to occur. For purposes of the DO analysis, the 30-day mean criterion will be used. The 30-day mean will be calculated from 4 weekly DO concentrations.

There are similar warmwater aquatic life criteria for DO. The warmwater criteria are generally 1 mg/L less than the equivalent coldwater aquatic life criteria. However, the application dates are different. The only DO criteria for the period of April 1 through September 30, the period shown on Figure 86, only apply if there are early life stages of warmwater aquatic life present. It should be noted that the definition of warmwater aquatic life also includes invertebrates, which tend to reproduce much more frequently than fish. Based on the likelihood of invertebrate reproduction, the 1-day minimum criterion for early-life will be used as a basis for comparison, but only for purposes of evaluating whether there should be a concern during impact assessment for the Program. For any actual evaluation of standards compliance, the State 303(d) list should be consulted. A summary of the Nebraska 303(d) list appears later in this appendix. The warm-water aquatic life DO criteria are only applicable to the site at North Platte.

To simplify the DO comparison of the data plotted on Figure 86 with appropriate aquatic life criteria, a summary is presented in Table 24. The DO concentrations are plotted against the values shown on the second (right) *y*-axis. As points of reference on Figure 86, 5 mg/L is appropriate for North Platte and the 6 mg/L is appropriate for all of the other sites, although there is no actual 5 mg/L gridline on Figure 86 and 6 mg/L would be below the DO coldwater criterion.

All of the sites downstream from Lake McConaughy show some results that are less than the respective DO criteria. The greatest number of samples (monthly averages actually) that have results less than the DO criterion for the sites are from the Sutherland site. If the actual cause was low DO in the release from Lake McConaughy, the greatest number would be expected at the Keystone site, which is located 1.5 miles downstream from Kingsley Dam. However, Lake McConaughy cannot be entirely ruled out, in that its releases contain high concentrations of oxygen-demanding substances, including sulfides and ammonia (Stansbury et al., 2002a – see later section on Lake Ogallala for details). In such a case, the full effect may not be fully shown until the water travels downstream, and particularly, if there are areas of flow in which turbulence is insufficient to provide aeration. If this is the case, then the effect of the oxygen demand are fully realized at the Sutherland site, and begin to decline with greater distance from the source.

Table 24. Summary of comparisons with aquatic life DO criteria for the mainstem of the North Platte River during the summer of 2001

North Platte River Site	Samples Used	Number < Crit.
at North Platte	23	3
near Hershey	20	8
near Sutherland	20	10
at Keystone	20	3
at Lewellen	20	6
at Bridgeport	20	0
at Mitchell	19	0
near Morrill	19	1
NE/WY State Line	19	0

In general, sites upstream from Lake McConaughy meet the DO criteria. Only the Lewellen inflow site to Lake McConaughy has a number of monthly averages less than the 6.5 mg/L criterion for coldwater aquatic life. Recall that the Lewellen site also was the warmest site of those upstream from Lake McConaughy. The DO results may in part be due to the warming of the water at the site. The only other site upstream from Lake McConaughy that did not meet the DO criterion was the site near Morrill. However, the rounded result at the time that the criterion was not met was actually 6.5 mg/L. To see that the DO concentration was less than the criterion, the monthly average DO concentration has to be reported to more than 1 significant decimal place. Strictly speaking, the DO criterion was met on the occasion that it was shown as not being met in Table 24.

The monthly NDEQ DO data from 2002 and 2003 at 4 sites on the North Platte River are plotted on Figure 87A. Because of the difficulty noted above in comparing monthly DO data with the DO criteria as described above, the percent saturation is also plotted on Figure 87B. The percent saturation is used based on the assumption that if the DO is near saturation, it should meet the DO criteria. However, the potential problem with that assumption ahs to do with the temperature from which the saturation concentration is estimated. If the temperature criterion is not met, the saturation concentration of DO may also be below the DO criterion.

The saturation DO concentrations were estimated using a nonlinear regression equation (Bleasdale Model). The equation was based on temperature and DO pairs develop on the USGS website, DO Tables (USGS, 2001). The elevation of each of the sites was retrieved from the USGS NWIS website. The station atmospheric pressure was calculated from the site elevations using a reciprocal exponential regression derived from a set of pressure and elevation pairs also retrieved from the USGS DO Tables website. An altitude adjustment based on the ratio of the station pressure to sea level pressure (760 mm Hg) was applied to each of the calculated DO saturation concentrations. The percent saturation was calculated from the ratio of the measured DO to the calculated saturation DO concentrations and plotted on Figure 87B.

The DO concentrations during the summer of 2002 are generally between 6 and 9 mg/L at all of the sites (Figure 87A). There were no measurement below 6 mg/L during 2002. The same

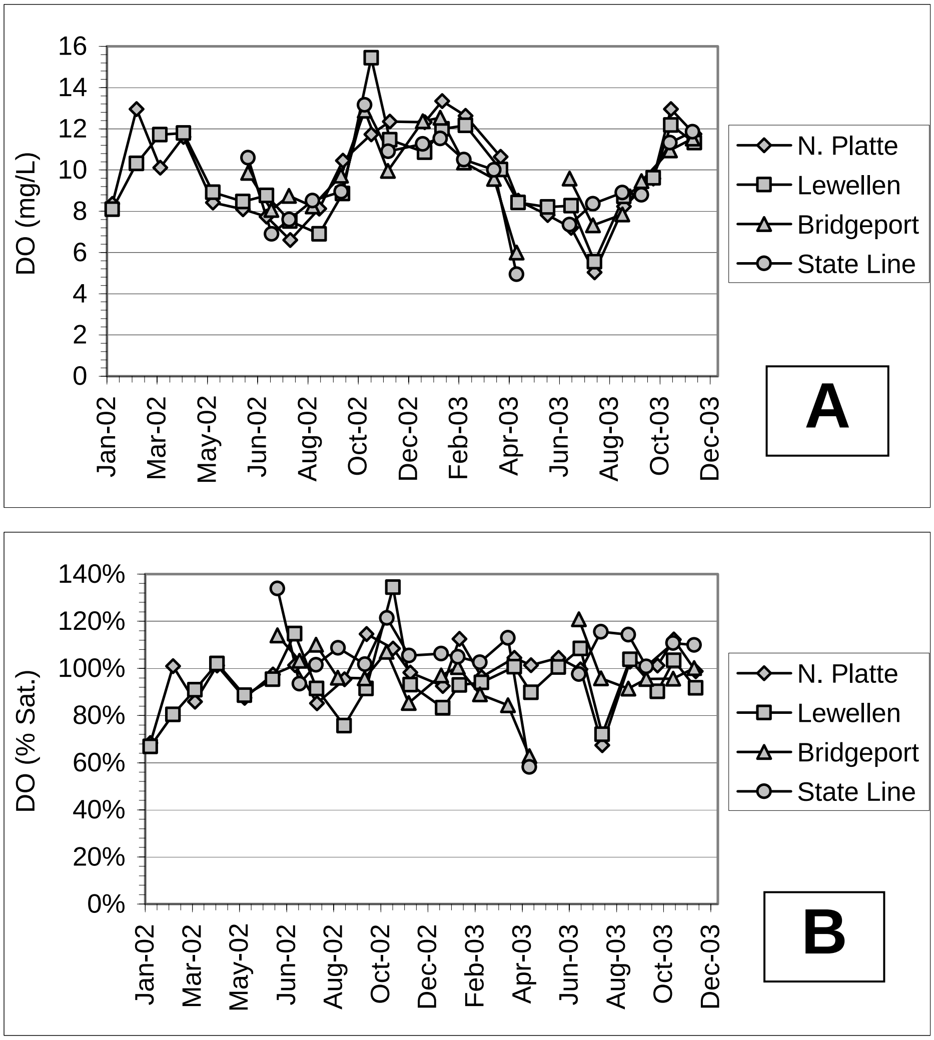


Figure 87. DO (A) and percent DO saturation (B) at 4 sites on the North Platte River in Nebraska during 2002 and 2003

cannot be said for the summer of 2003. In April 2003, the DO concentration fell below 6 mg/L at the sites at the State Line and Bridgeport. There were no measurements the following month. During August 2003, the DO concentration fell below 6 mg/L at the other 2 sites, Lewellen and North Platte. Late summer is the more likely time to have low DO concentrations, because the water is warmest and the saturation DO concentration is lowest at that time of the year. In addition, Lewellen and North Platte are the 2 farthest downstream sites on Figure 87, where the warmest temperatures are expected.

The majority of the DO measurements plot around the 100 percent saturation gridline on Figure 87B. As expected, the lower DO concentrations shown on Figure 87A plot below the 80 percent saturation gridline. However, there are several other DO measurements that also plot below the 80 percent saturation gridline, including the first 2 samples shown in January 2002. Between January and June 2002, measurements were only made at 2 of the sites, Lewellen and North Platte. The DO saturation at the 2 sites fell to near 60 percent at those sites during January 2002 (Figure 87B), although the DO concentration at the time was approximately 8 mg/L (Figure

87A). The one other time that there was a DO measurement below 80 percent saturation was at Lewellen during August 2002, when the DO concentration was 7 mg/L (Figure 87).

Effects of Program Alternatives

As was done for the North Platte River in Wyoming, the alternatives comparison for the North Platte River in Nebraska will be based on their effects on TDS. The alternatives comparison will be based on the regression relationships developed above based on TDS and flow at the Lisco gage and shown on Figure 76. Although the relationships will be based on data from the Lisco gage, they will be applied to the modeled flows at Lewellen.

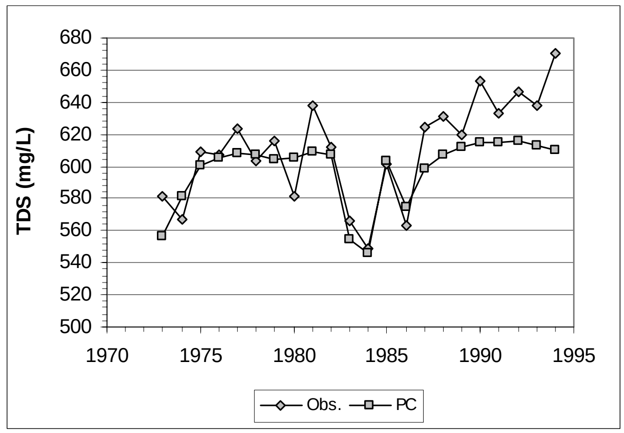


Figure 88. Present Condition TDS (from seasonal regressions) of the North Platte River at Lewellen and historic TDS at Lisco, Nebraska

Figure 88 shows a comparison between the measured annual average TDS at Lisco and the calculated TDS for the Present Condition at Lewellen. There is much less variation in the Present Condition TDS than there is in the measured data, but the Present Condition TDS shows relatively good agreement with the measured TDS, particularly between the water years of 1973 (the first year with measured data) through 1987. The two sets of TDS data begin to diverge beginning in 1988. In the period of 1988 through 1994, the Present Condition TDS is consistently lower than the measured TDS. During this period, the measured TDS shows an increase over what had been observed earlier in the TDS data record. Although the number of samples decreased continually over the period of record that the TDS was sampled at the Lisco gage, *i.e.* from over 30 per year in the early 1970s, to monthly samples in the later 1970s through most of the 1980s, to 6 samples per year in the late 1980s and early 1990s, and to only 4 samples in 1994, the decline in samples does not appear to be a factor in the increase in TDS at the end of the measured data set. An evaluation was investigated to see if higher TDS values were being unduly weighted within a year as the number of samples declined. This did not appear to be the case.

Figure 89 shows a comparison of the annual average TDS of each of the alternatives with that of the Present Condition. At the scale of the plots, any differences are virtually indistinguishable. In the case of the Governance Committee and Full Water Leasing alternatives, the TDS plots are essentially overlain on that of the Present Condition. There are some periods when there is a small separation between the time series plots of the mean annual TDS of the Present Condition and the TDS of the Water Emphasis and Wet Meadow Emphasis alternatives. These separations represent decreases, which occur at higher TDS concentrations during drought years.

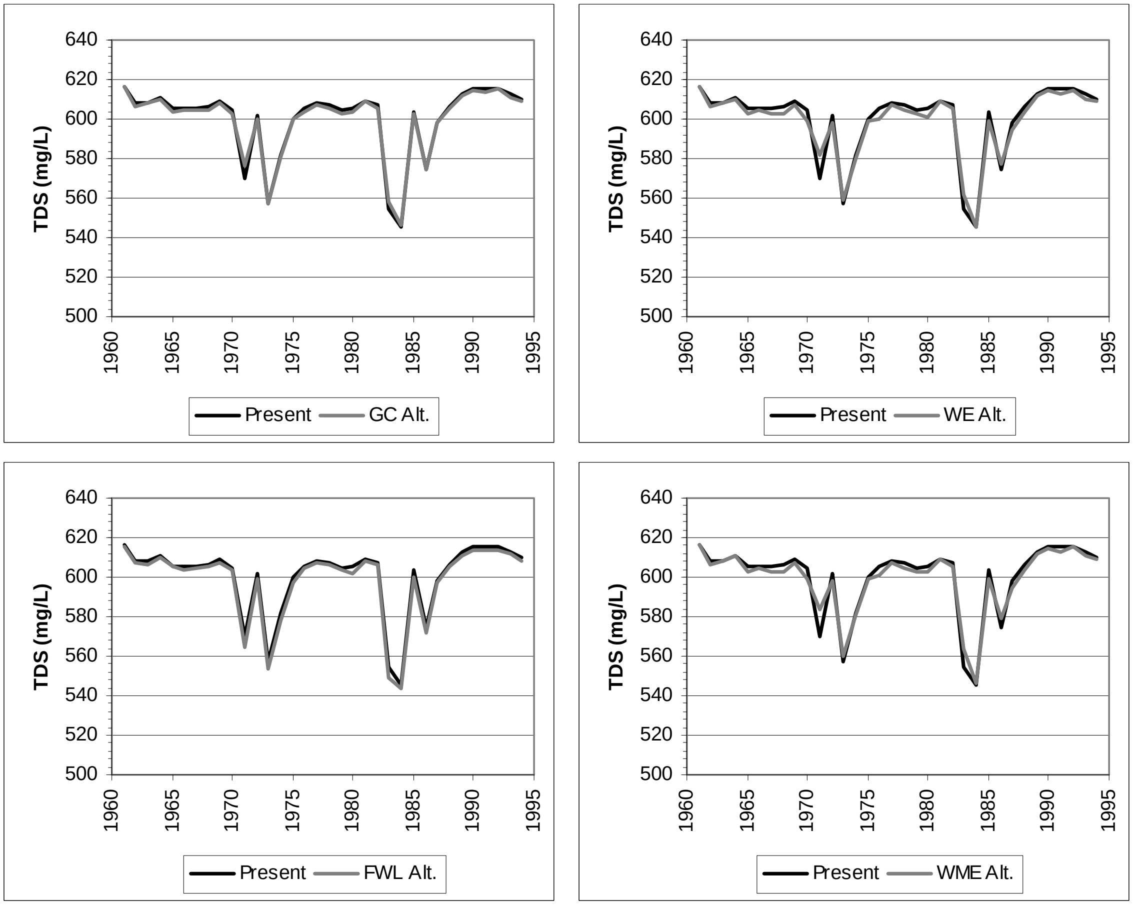


Figure 89. Mean annual TDS of the North Platte River at Lewellen, Nebraska, under the Present Condition and each of the 4 Program alternatives based on seasonal regressions

Averages tend to mask variation in data sets. The monthly data do show some differences at time. The total range of differences in TDS based on the monthly data are summarized in Table 25. The net change reflects the average of the differences of the TDS of each alternative from that of the Present Condition. All of the alternatives show a net decrease in TDS relative to the Present Condition. However, the differences are extremely small with the greatest decrease at only 2 mg/L, a value that is immeasurable, given the degree of error in a gravimetric TDS determination.

Table 25. Range of monthly changes in TDS in comparison with the Present Condition

Alternative	Maximum Decrease	Net Change	Maximum Increase
Governance Committee	-22	-1	58
Water Emphasis	-64	-1	140
Full Water Leasing	-63	-2	24
Wet Meadow Emphasis	-61	-1	140

In general, the decreases in TDS in the North Platte River occur in September. All of the larger decreases occur in September. Alternatively, the increases, particularly the larger increases, occur in the spring, *i.e.* April through June. There are some small increases during the summer, but for the most part there is no change in TDS relative to the Present Condition during the summer and winter months.

North Platte Tributaries, Nebraska

Based on comments related to fisheries in North Platte tributaries, particularly in relation to the Full Water Leasing Alternative, flow and water quality data were retrieved from the USGS NWIS database for sites tributary to the North Platte River in Nebraska. There were flow data on a number of drains. However, water quality data were much more limited. Although there is still the issue of how much water would (or could) be leased in the area, a review of the data might provide some insight, maybe enough to make a generic assessment that should be good enough for a programmatic EIS.

USGS Data

There are 2 drains with long-term flow records, the Gering Drain near Gering and the Ninemile Drain near McGrew. The daily flow records for the drains are plotted on figures 90 and 91. The flow of the Ninemile Drain is much larger than that of the Gering Drain. The flow of the Ninemile Drain is centered on the 100 ft^3/s line on the plot (Figure 91); actually, the median flow of the Ninemile Drain is 101 ft^3/s. The flow of the Gering Drain is centered below the 100 ft^3/s, and its median flow is 31 ft^3/s. However, the actual flows in the drains are immaterial to the question. The percentage of flow that originates from irrigated agriculture is the real value that may provide an insight into answering the question at hand.

The annual patterns of flow in both of the drains on figures 90 and 91 resemble what is known in hydrology as a recession curve. A recession curve is used to determine the base flow of a stream. The idea is to take the stream flow data at a peak and plot the decline until the flow levels off, at which point the flow contribution from ground water dominates. The pattern represents an initial contribution from surface runoff, followed by the drainout from soil water, and finally the contribution from alluvial recharge. Irrigation is seasonal. Consequently, the contribution to the drain from irrigation return flows, including drainage will decline following the irrigation season. The base of the annual curves on figures 90 and 91 should then represent a contribution from ground water sources other than irrigation, at least theoretically (or hypothetically). In this case, the contribution from irrigation could hypothetically be related to a short term contribution from rainfall in terms of the annual hydrograph. The large spikes in flow on figures 90 and 91 are probably contributions from storms. These spikes would only contribute to flow in a short term, *i.e.* several weeks or so.

There were 2 other drains that were gaged in the early 1960's. The daily flow data from those gages, along with the flows from the Gering and Ninemile drains from the same period are plotted on Figure 92. All of the drains show the same general pattern in their annual hydrographs, once again resembling a recession curve. All of the drains that contribute to the North Platte River drain the sandhills, at least in part. The sandhills extend to the South Platte River (see the analysis of the Tamarack proposal for an example). The sandhills would contribute base flow to all of the drains. The low flows of the annual hydrographs on Figure 92 would hypothetically represent the flows in the streams if all irrigation contributions were eliminated. Those low flows should therefore bracket the effects of the Full Leasing Alternative.

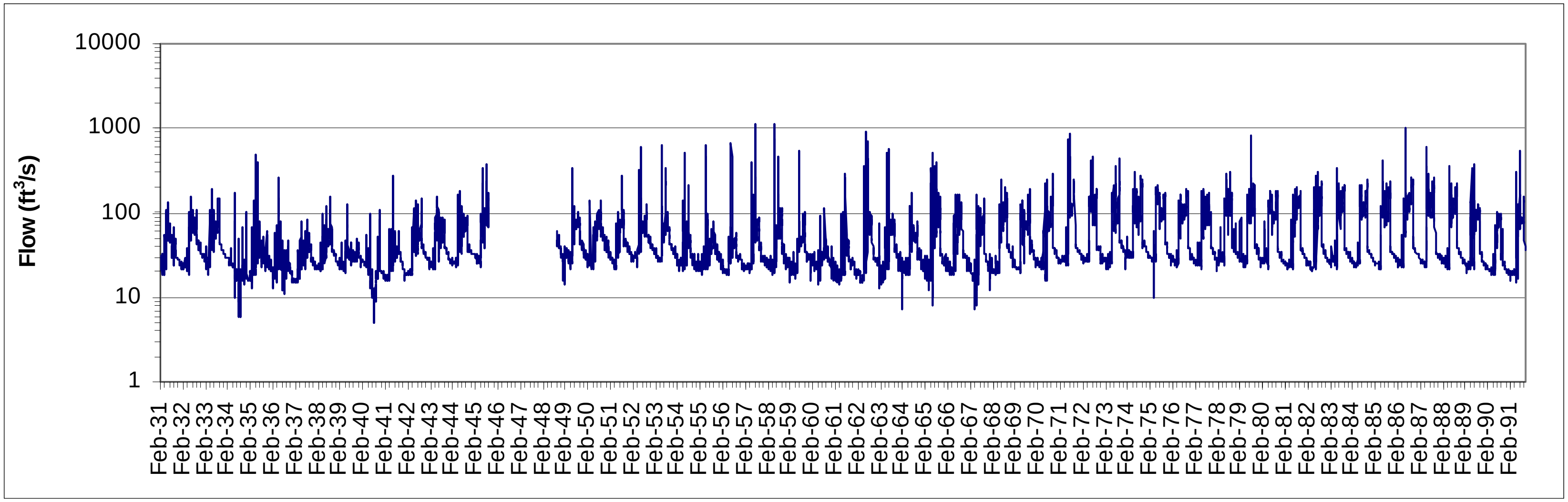


Figure 90. Daily flow of the Gering Drain near Gering: water years 1931 through 1991 - 1946-1948 are missing

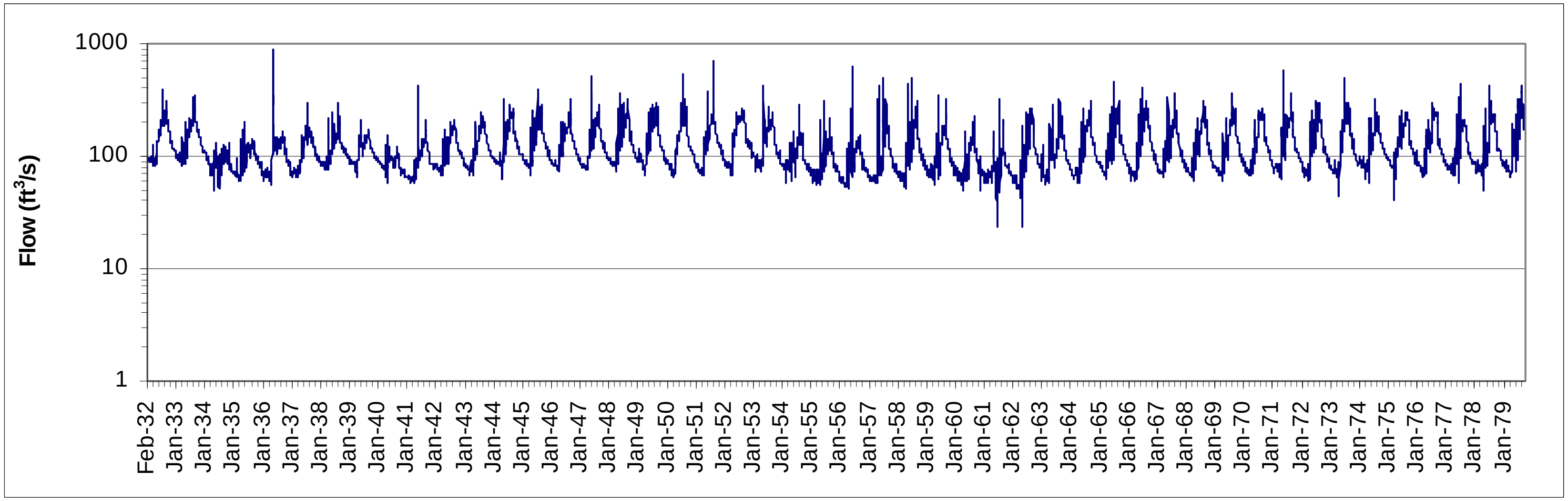


Figure 91. Daily flow of the Ninemile Drain near McGrew: water years 1932 through 1979

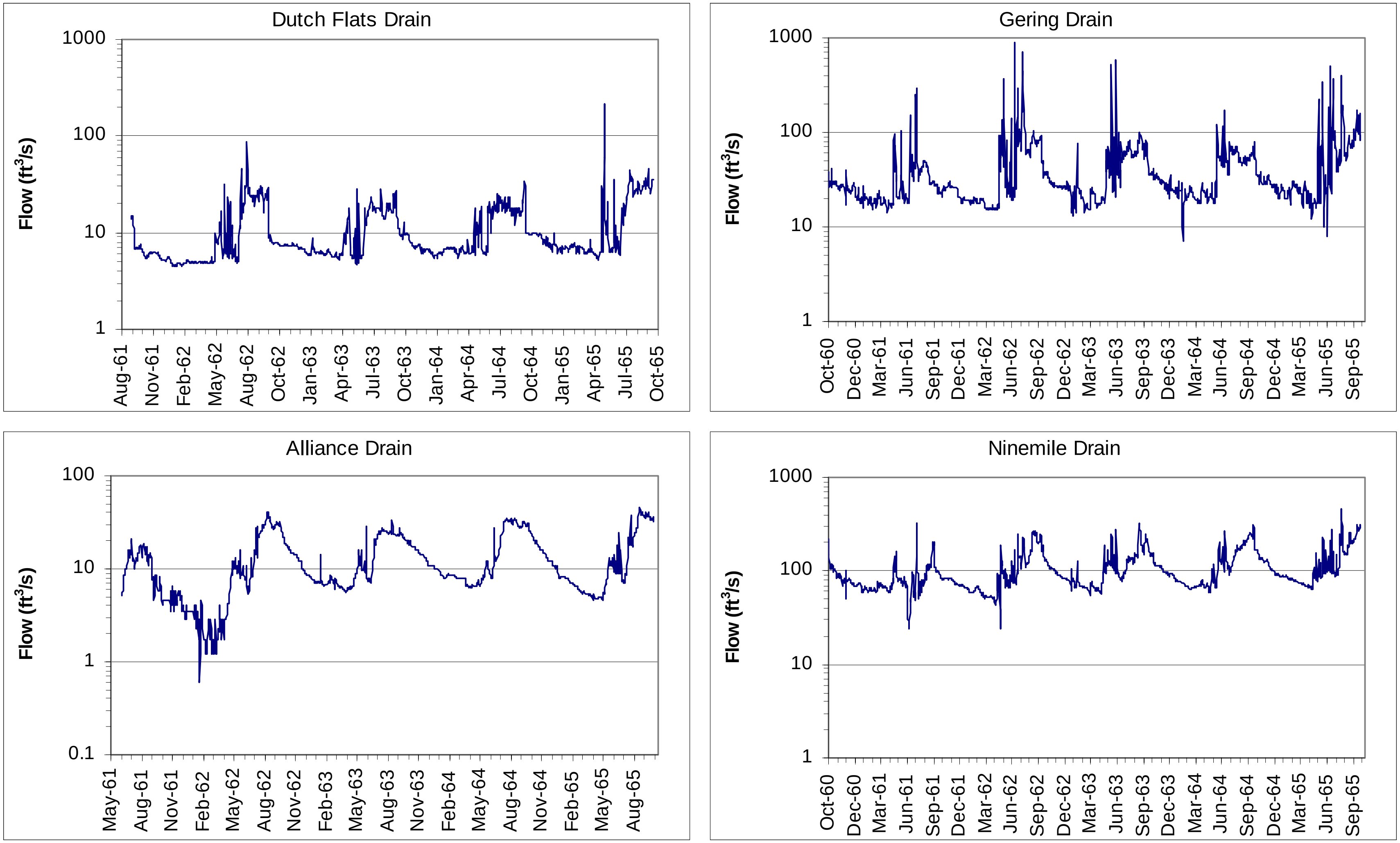


Figure 92. Flows in 4 drains tributary to the North Platte River in Nebraska during water years 1961 through 1965

The low flows in the drains on Figure 92 vary from year to year. If we accept the above premise, the variations in low flows represent the variation in ground water recharge in wet and dry years. This variation does not affect the analysis, but does indicate that there can be no absolute answer to the question, at least in terms of flow.

In addition to the 2 drains plotted on Figure 92, there was also another gage site on Ninemile Drain during water years 1961 through 1965. The daily flows on the 2 sites on Ninemile Drain are shown on Figure 93. Although the flow in the Ninemile Drain near McGrew are much

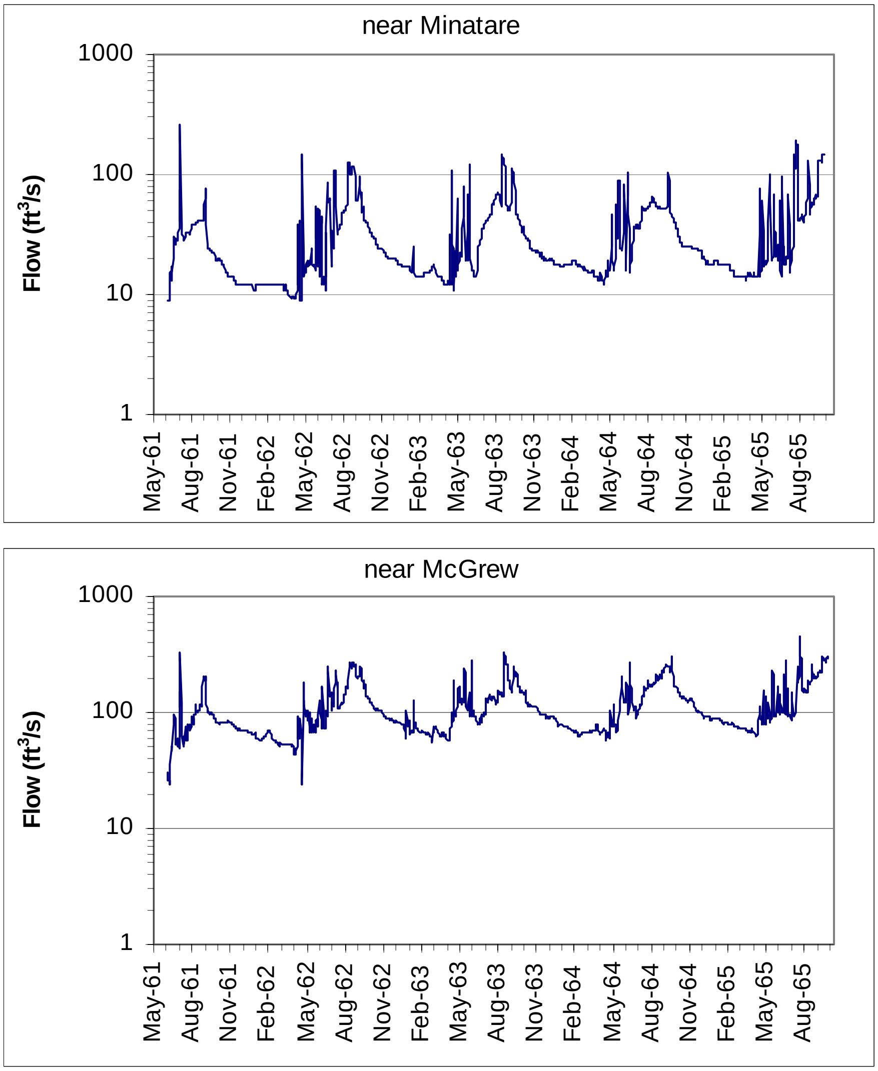


Figure 93. Daily flows at 2 gages on Ninemile Drain during water years 1961 through 1965

higher than those near Minatare, the patterns of flow in the hydrographs are remarkably similar. The difference represents gains in flow between the 2 sites, The similarity in the hydrographs would indicate that the gains are from similar sources. Hypothetically, the gains would be from irrigation returns, large storms, and ground water, all of which contribute in a similar manner between the sites.

The relationship between flows at the 2 gages is shown on Figure 94. The regression of the downstream flow on the upstream flow is curvilinear. The gains show a drop off at higher flows. The differences at higher flows would apparently represent a difference in the magnitude of gains due to surface runoff. This would not be an effect of ground water gains. However, this does not contribute particularly to answering the question concerning the effect on flow due to the implementation of the Full Leasing Alternative.

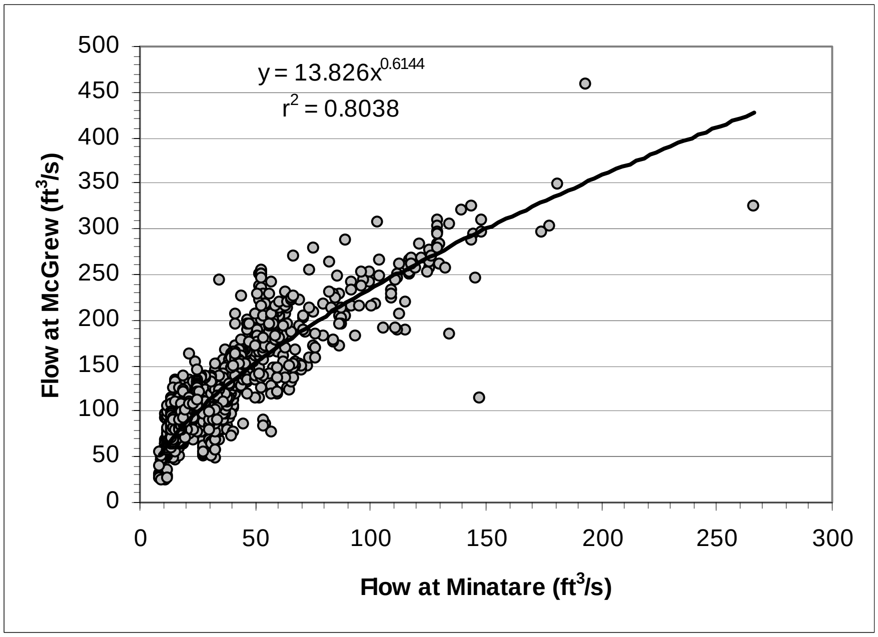


Figure 94. Relationship between flows in the Ninemile Drain at the Minatare and McGrew gages during water years 1961 through 1965

Figure 95 explores low-flows in more detail in the Gering Drain which has the longest-term flow record of any of the drains. Figure 95 shows a cumulative frequency distribution of annual minimum flows, along with a similar distribution of 7-day low flows. The reason for including the 7-day low flows has to do with the low-flow spikes (those dropping below others for a day or so on Figure 91). The low-flow spikes appear to be errors in the gage record, potentially due to an accumulation of sand in the stilling wells or the activities of beavers or muskrats. Anyway the 7-day low flow could eliminate those factors.

The distribution of flows on Figure 95 represents the equivalent of an inverse flood study. Where a flood study uses maximum annual flows, Figure 95 uses low flows. The frequencies therefore represent annual probabilities of low flows. If the low-flows represent base flows, as hypothesized above, then the low flows on Figure 95 would represent those base flows. The frequencies on the ordinate of Figure 95 would represent probabilities of a flow being that low in the absence of irrigation contributions. Based on Figure 95, the flow (7-day) would be 15 ft^3/s or less in about 15 percent of the years. Conversely, the low flow would be greater than 15 ft^3/s in 85 percent of the years.

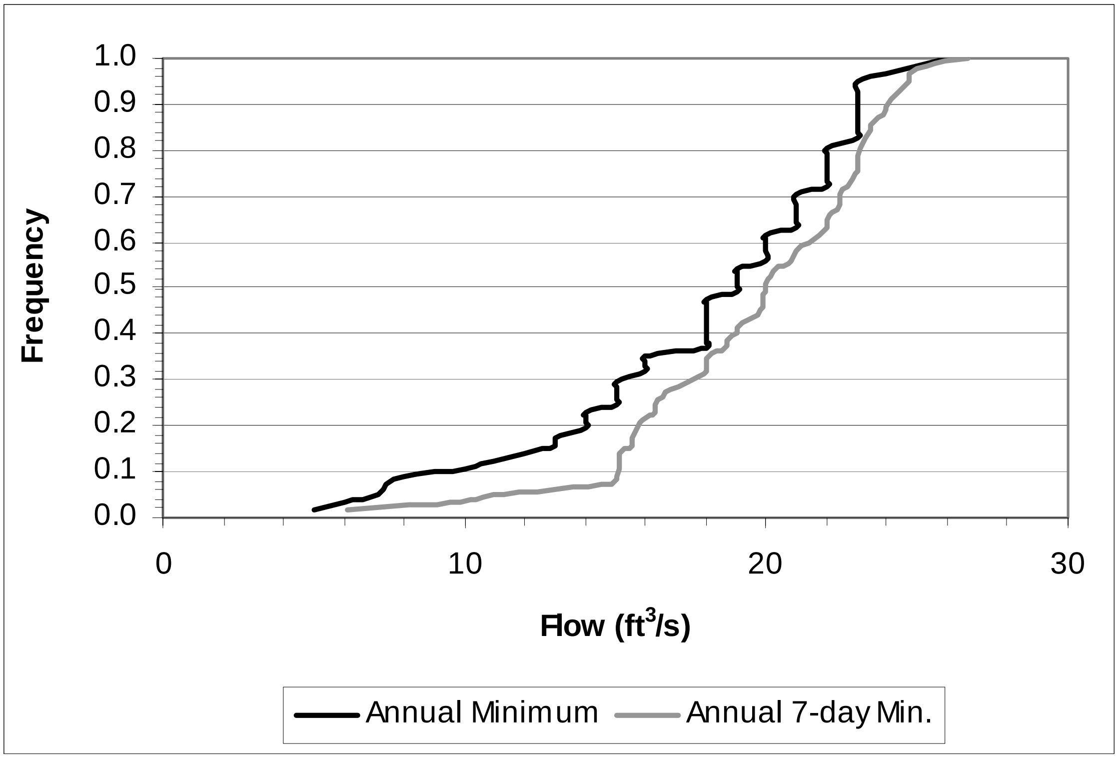


Figure 95. Cumulative frequency distribution of annual low flows in the Gering Drain

In flood studies, a recurrence interval can be calculated from the frequencies. The recurrence interval is simply the reciprocal of the frequency. The conversion to recurrence intervals for the Gering Drain is shown on Figure 96. Figure 96 indicates that flows in the Gering Drain less than the 15 ft^3/s low flow could be expected about every 15 years or so, and low flows of 10 ft^3/s or less could be expected every 25 years. It should be noted that the data on Figure 96 are not exactly as sophisticated as what is done for a flood study. The usual procedure for a flood study is to fit the data to a distribution that allows extrapolation beyond the period of record of the data. Nothing like that has been attempted here, but Figure 96 does represent the 57 years of data that went into it.

The 15 ft^3/s low flow that was flagged in the discussion of figures 95 and 96 is approximately ½ the median flow of the Gering Drain. The low-flow in the drain obviously has been lower than that in the past. The question is, what is the effect that such low flows had in the past. That would translate into something of an assessment of the effects of the Full Leasing Alternative under a worst case condition for the Gering-Mitchell and Tri-State irrigation districts drains.

Figures 97 and 98 show plots of equivalent data for the Ninemile Drain as was shown on figures 95 and 96 for the Gering Drain. Figure 97 indicates that flows of 56 ft^3/s or less could be expected in less than 10 percent of the years. As above, the 56 ft^3/s is approximately ½ of the median flow of the drain over the 47 years of record. The recurrence interval for flows less than 56 ft^3/s is 15 years or more (Figure 97).

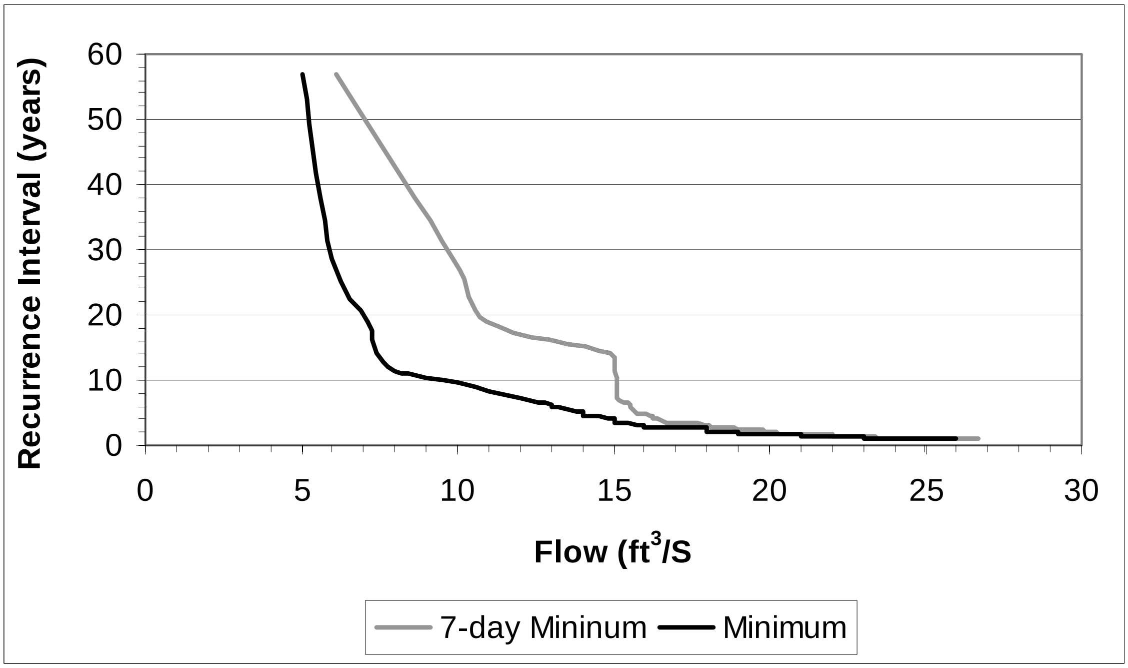


Figure 96. Low-flow recurrence intervals in the Gering Drain

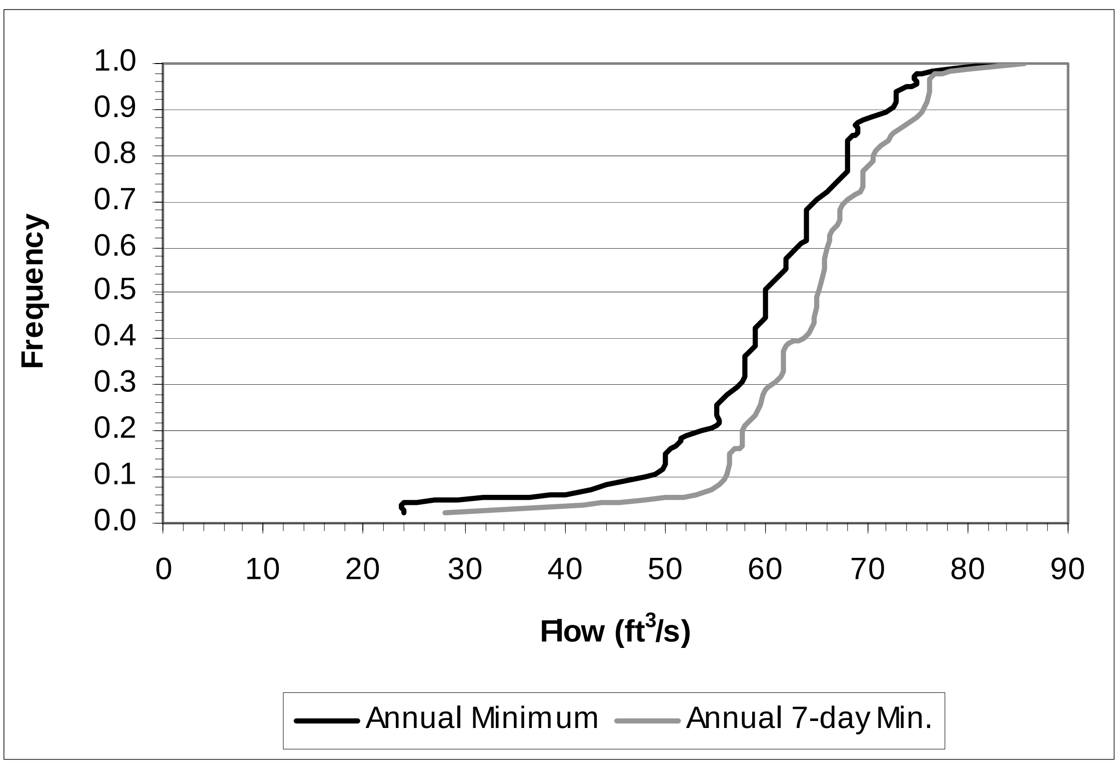


Figure 97. Cumulative frequency distribution of annual low flows in the Ninemile Drain

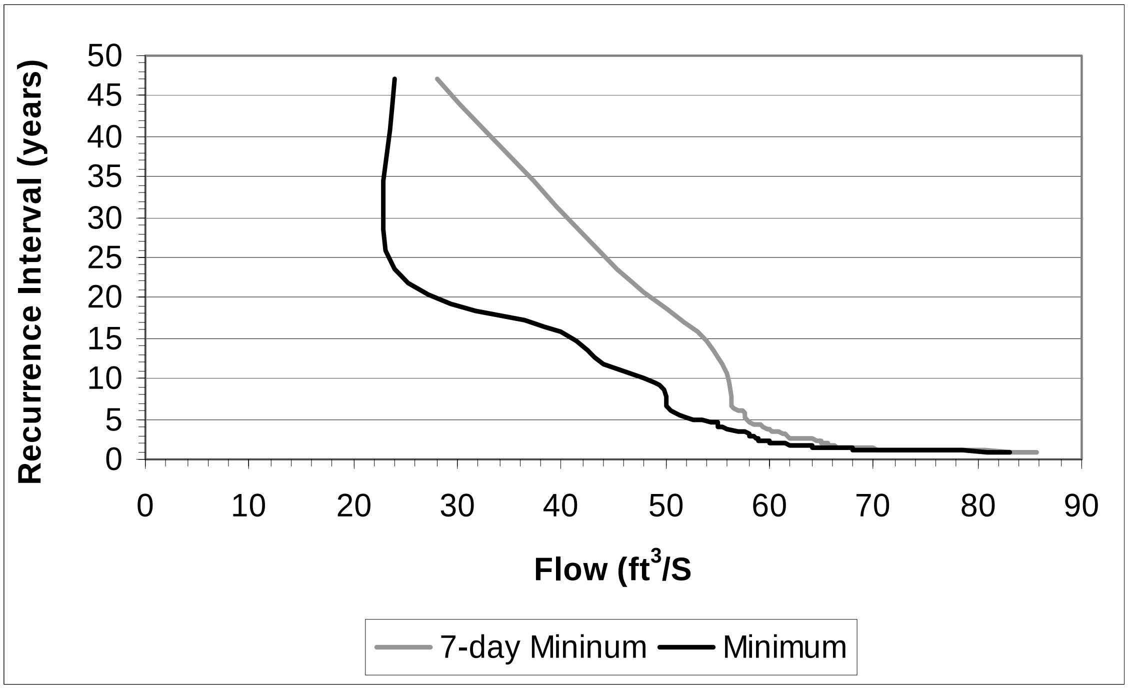


Figure 98. Low-flow recurrence intervals in the Ninemile Drain near McGrew

A number of stream characteristics can be related to flow using power functions. There are limited water quality data for the Gering and Ninemile drains. Included in the data are measurements of flow and gage height. These data are used to derive power relationships of gage height to flow on Figure 99. This could put the flow data into some perspective that could ultimately relate to fish habitat. For example, at the lowest flows evaluated above for the 2 drains, the water would be over 1 foot deep in each of the drains. Alternatively, even though the relationships between depth (gage height) are exponential, the response (change in depth) to changes in flow is not that great (Figure 99). In each of the drains, increases in flow of 150 ft^3/s only result in about a 1.5 foot increase in depth. Consequently, small changes in the low flows may not have a great effect on fish habitat. Alternatively, if the peak flows are due to storm runoff, reduction in irrigation return flows would not greatly affect those. The reduction due to the elimination of irrigation would be at the intermediate level flows in the annual hydrograph. Can we assess what affect that would have on the fisheries?

There is also the issue of temperature. The irrigation returns from the drains appear to augment the flows in the summer. Consequently, the return could buffer against rises in temperature. Alternatively, the application of water for irrigation could also be a factor in warming the water.

The above referenced water quality data included some temperature measurements. These are summarized on Figure 100. Only 2 of the drains with water quality data are included among those with gage records. There are only 3 temperature measurements for the Gering and Ninemile Drains, the 2 drains with both gaged flow and water quality data. As can be seen on Figure 100, upper limb of the confidence interval for the temperature data of the Ninemile Drain is somewhat above the 22°C (72°F) Nebraska coldwater aquatic life criterion at 24°C (75°F). However, the limit that was used in the 1990 Lake McConaughy study (Yahnke, 1990) was 75°F

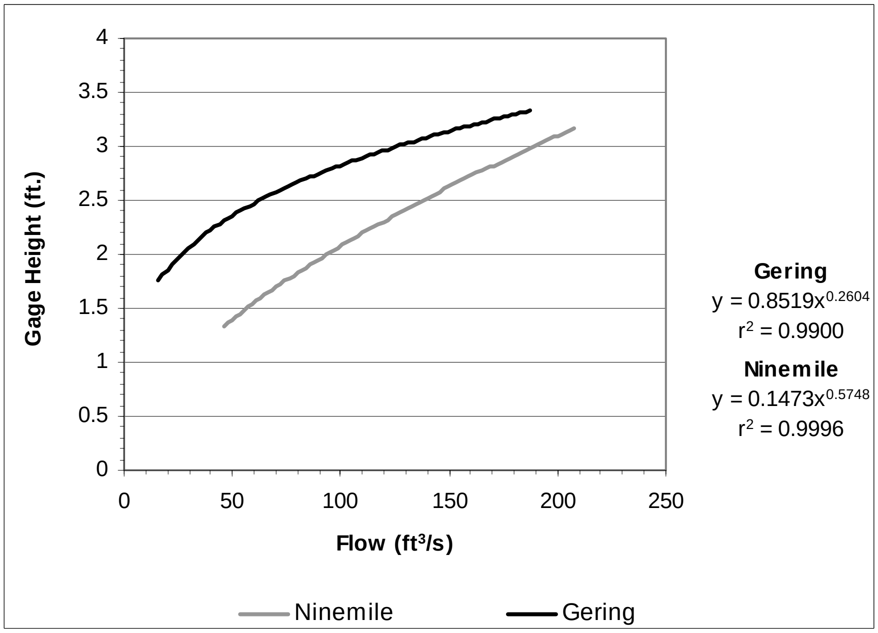


Figure 99. Relationship between gage height and flow in the Gering and Ninemile drains

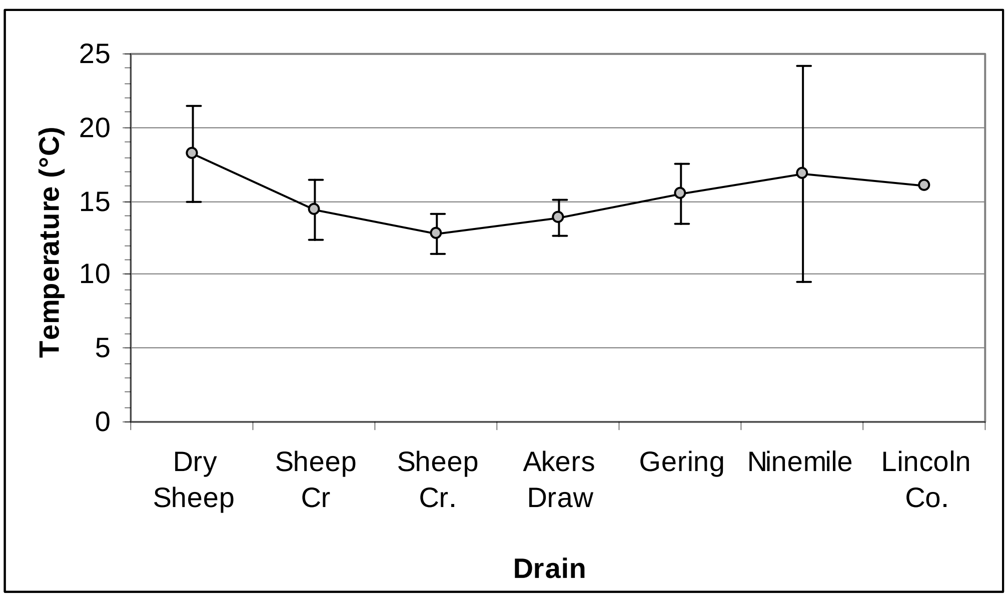


Figure 100. Temperature of 7 drains tributary to the North Platte River in Nebraska

(23.9°C). Of the 3 measurements in the Ninemile Drain, the maximum temperature, which was measured in July, was 23°C (73°F). If the trout in the drains are remnants of the Lake McConaughy population, then the high temperature is probably not a problem. The Lake McConaughy strain of rainbow has been planted all over the Intermountain West because of their tolerance to high temperatures, at least high for trout. So the only remaining question relates to the removal of the irrigation returns from the drains. If it has historically raised the temperature, then conditions should improve. Alternatively, if the return flows have been a buffer, then temperatures would rise and potentially cause some loss of trout. Even so, where either native ground water or subsurface irrigation drain water enters the stream, there should be cool water refugia that would tide over some of the fish.

This analysis assumes a complete removal of the returns. The Program formulation of the Full Leasing Alternative contemplates that the water supply in the area would be depleted by at most 32 percent. Under those circumstances, the worst case analysis presented above would not apply. The effects would be considerably less severe.

NDEQ Data

Temperature and DO data collected by the NDEQ from the North Platte River at several sites are described elsewhere in this appendix. In addition to those sites, the NDEQ sampled a number of tributaries to the North Platte River in 2001, 2002, and 2003. The sites are listed in Table 26. As noted in the footnotes to Table 26, the Gering Drain was sampled at 2 sites in 2002. Because the samples are intermingled in time, the data are only being put to limited use. Although flows at McGrew could be estimated from the regression shown above, water quality at the 2 sites appears to be very different. One further note on the sites in Table 26 – the Ninemile Drain discussed above is also known as Ninemile (and Nine Mile) Creek, its name in Table 26.

Table 26. Nebraska Department of Environmental Quality sample sites during 2001 through 2002

2001 Sites	2002 Sites	2003 Sites
Pumpkin Cr.	Pumpkin Cr.	Pumpkin Cr.
Red Willow Cr.	Red Willow Cr.	Red Willow Cr.
Ninemile Cr.[1]	Ninemile Cr.[2]	Ninemile Cr.[1]
Gering Drain	Tub Springs	Tub Springs
Winters Cr.	Winters Cr.	Winters Cr.
Horse Cr.	Blue Cr.	—
Birdwood Cr.	Otter Cr.	—

[1] McGrew
[2] McGrew and Minatare

According to Nebraska Game and Parks Commission (2005), Dry Sheep, Dry Spotted Tail, Sheep, Winters, Spotted Tail, Nine Mile, Stuckenhole, and Red Willow creeks and Tub Springs support brown and rainbow trout fisheries. Of these water bodies, only Red Willow, Ninemile, and Winters creeks and Tub Springs appear in Table 26, but information on the Sheep Creek sites was presented above. Consequently, Red Willow and Winters creeks and Tub Springs will be the focus of this section of the report. The location of the NDEQ water quality sites is shown on Figure X12. All of these creeks are classified for coldwater aquatic life by the NDEQ. The temperature criterion for coldwater aquatic life is 22°C (72°F).

As shown in Table 26, Red Willow and Winters creeks were sampled in 2001, 2002 and 2003. Tub Springs was only sampled during 2002 and 2003. There were differences in the sampling frequency and sample procedures among the 3 years. In 2001, samples were collected weekly from April through September. In 2002 and 2003, samples were collected monthly at each of the sites. In 2002, Red Willow and Winters creeks and Tub Springs were sampled from January through December, while the other three sites were only sampled from July through December. In 2003, all sites were sampled in all months. The limited sampling in 2002 is another reason for limiting this section to the three sites listed above. Ninemile Creek was also sampled in all months in 2002 and 2003, but as noted above, at two different sites. Although Ninemile Creek

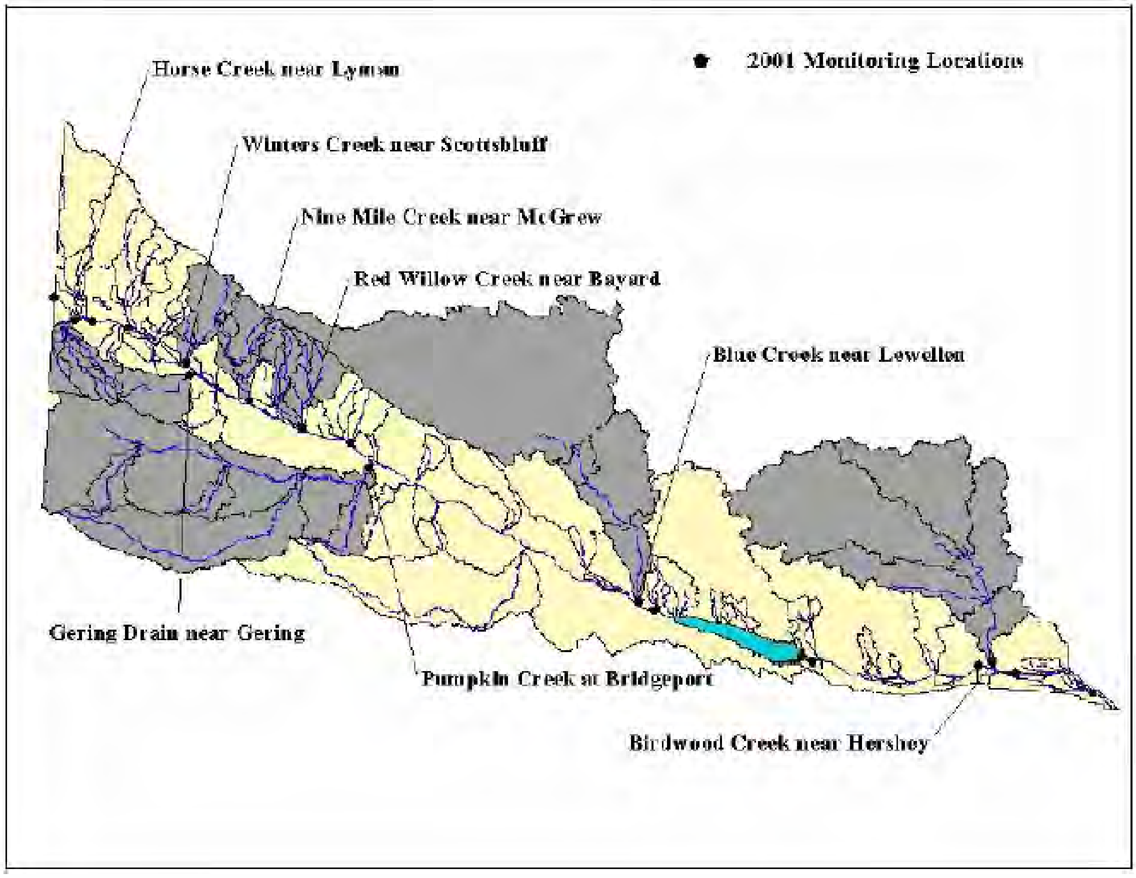


Figure 101. NDEQ Tributary Monitoring Locations in the North Platte Basin

and the Gering Drain will not be the focus of this section, some of the data will be presented to supplement the flow analysis that was presented above.

Red Willow Creek was also sampled at two different sites, both of which are near Bayard. The difference in the two sites is not as well defined as was the case for the two sites on Ninemile Creek. The winter-spring samples in 2002 were collected from the northeast (NE) site, while all other samples were collected from the southeast (SE) site.

Winters Creek is the only tributary that was sampled at the same site on all occasions in 2001 and 2002 and at the same site in both years and those data will be presented first. Because of its consistency, Winters Creek will provide a basis for comparison for the other sites, where sampling was not as consistent.

Figure 102 shows the temperature and EC data from Winters Creek. Temperature is obviously critical for coldwater fisheries, while EC may serve as an indicator of drainage inflows. Ground water from the Sandhills aquifer is relatively dilute, with an EC generally less than 300 µS/cm. Alternatively, the EC of the North Platte at Whalen Diversion Dam, which is the primary source of the irrigation supply, is generally between 500 and 1000 µS/cm (Figure 2) and would be further increased by evapotranspiration by crops. The EC should be higher when the flow of the creek is dominated by drainage flows.

Figure 102 shows a trout temperature criterion of 68°F. It was noted above that the Lake McConaughy strain of rainbow trout was relatively tolerant of warm water and a temperature criterion of 75°F was used in previous studies. However, brown trout are not as tolerant of warm water as rainbow trout in general, and in particular, relative to tolerant strains of rainbow trout. Because of the presence of brown trout, a lower temperature criterion than was indicated earlier is applied to Figure 102.

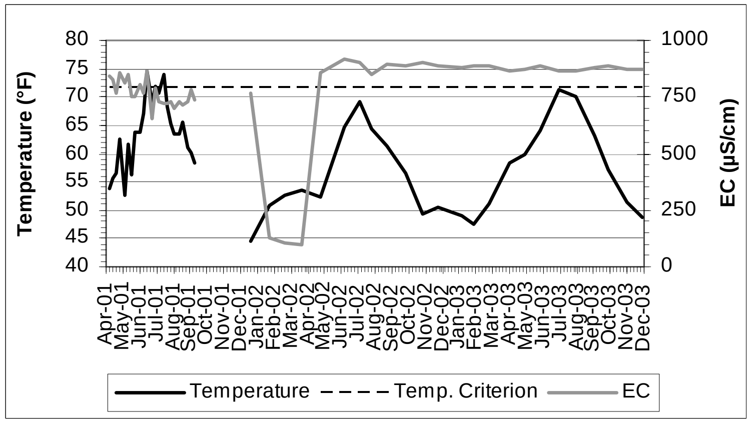


Figure 102. Winters Creek temperature and EC during 2001 through 2003

The weekly EC and temperatures in 2001 show numerous spikes and decreases. Over the sampling period, temperature shows a typical seasonal pattern, while EC shows a general decrease from near 900 to about 750 µS/cm at the end of the summer. This is the opposite of what is usually expected with drain water inflows. The peak temperature is just below 75°F in early July. The temperature dropped below the temperature criterion through much of July 2001, but once again increased above the criterion at the end of the month. The temperature in January 2002 was at its minimum, but in the mid-forties, which is indicative of ground water inflow (Figure 102). The EC at the time was about the same as what had been observed at the end of the irrigation season, as represented by the sample at the end of September. In February, the EC dropped precipitously to near 100 µS/cm and remained there until May. The temperature remained in the low-fifties throughout that period and beyond, *i.e.* into June, when the EC jumped to near 900 µS/cm. The EC remained around 900 µS/cm throughout the remainder of 2002 and throughout 2003, while the temperature exhibited typical seasonal patterns. The peak temperatures occurred in July at about 69° in 2002 or just above the temperature criterion and at just over 71° in 2003.

Another factor that can be considered is flow. As noted above, there are no flow data associated with the 2001 water quality data, but there are flow data in 2002 (Figure 103). The 2002 gage height readings for the Winters Creek monitoring site are also shown on Figure 103. The flow decreased from 44 ft^3/s in January to 15 ft^3/s in mid-July. In August, the flow increased dramatically to over 90 ft^3/s, following which it fell back to where it had been in the winter. The hydrograph shown on Figure 103 is more typical of a stream that is

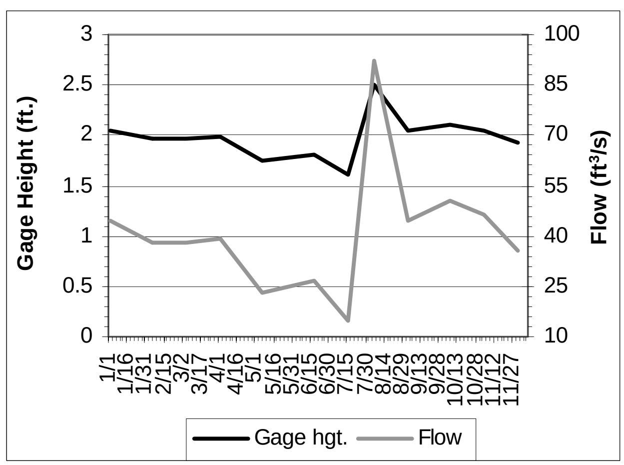


Figure 103. Winters Creek gage height and flow during 2002

affected by diversions, more than by drain flows. This is particularly so because the peak is not accompanied by any change in EC, which remained relatively constant following its increase several months earlier. There should also be an increase in flow in the spring, when the ground thaws. If the stream is being fed by the Sandhills aquifer, the spring peak should be relatively rapid following the thaw. The decrease in the spring is typical of what would occur with diversions for irrigation

The gage readings do not show a very large range, *i.e.* less than 1 foot (1.6 to 2.5 feet), despite a range of flow from 15 ft^3/s to over 90 ft^3/s. This is a typical of gage height-flow relationship. As noted in the previous section, the relationship should fit a power function, which it does, *i.e.* $r^2 = 0.99$, as on Figure 99 for the Gering and Ninemile drains.

All of the above indicates a set of complex interrelationships that are atypical of what would be expected of a stream in the area. Part of the problem is that 2002 was an atypical water year. In the North Platte Basin, 2002 was part of a very severe drought condition. The lack of a spring peak in the stream flow may be more a reflection of the drought condition than any factor related to irrigation.

The Tub Springs samples were also all collected at the same site, although measurements were only made during 2002 and 2003. The available temperature, EC, gage height, and flow data are shown on Figure 104. There is considerable similarity in the temporal distribution of temperature and EC between Tub Springs and Winters Creek (compare figures 102 and 104). In both streams, the January temperature was in the mid-forties and rose to a peak of about 70° in July. The EC in both streams was near 800 µS/cm in January and decreased precipitously in February, to about 100 µS/cm in Winters Creek, but to about 200 µS/cm in Tub Springs. Both streams showed a relatively constant EC through the summer and fall. However, there was a large difference in the distribution of EC in the spring. Where the EC remained constant between February and May in Winters Creek (Figure 102), there was a large spike in EC in March in Tub Springs (Figure 104).
In the months surrounding the EC spike in Tub Springs, the EC was between 100 and 200 µS/cm.

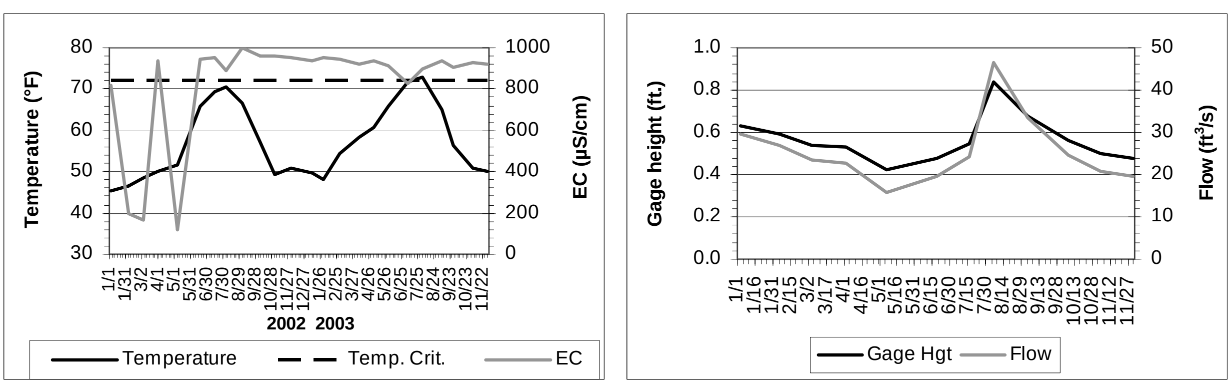


Figure 104. Tub Springs temperature and EC in 2002 and 2003 and gage height and flow during 2002

There is also considerable similarity between the flow patterns (and gage heights) over the course of 2002, although the flow in Winters Creek is about twice that in Tub Springs. Both show a general decrease in flow from January through July, although the decline is interrupted by a small increase in flow in June in Winters Creek (compare figures 103 and 104). The one thing that both hydrographs share is a peak in flow in August, followed by a drop in September. The decrease in flow during the fall is more continual in Tub Springs than in Winters Creek (once again, compare figures 103 and 104).

The similarities are exemplified by the correlations shown in Table 27. All of the variables, except EC are highly correlated between the two sites, *i.e.* probability of a greater r occurring by chance alone is less than 1 in 100. The best correlation is between the temperature distribution at the 2 sites. This is not particularly surprising in that the 2 sites are not far apart geographically and water temperature is mostly controlled by insolation (incident solar radiation), which should be similar in nearby sites. However, the correlation between flows at the two sites also indicates a similarity in the controls on flow. Because Tub Springs has a predominant ground water source, the implication is that Winters Creek does as well. The ground water may originate from seepage from the natural water table, canal leakage, deep percolation from irrigation, or a combination of all or some of these, the last of which is most likely.

Table 27. Correlations between EC, temperature, gage height and flow in Tub Springs and Winters Creek

EC	*Temp*	*Gage Hgt.*	*Flow*
0.538	0.954	0.771	0.830

The last of the tributaries that will be a focus of this section of the appendix is Red Willow Creek. Recall, however, that the data from the winter and early spring of 2002 originate from a different site from those collected in 2001 and the remainder of 2002.

Figure 105 shows the temperature and EC in Red Willow Creek during 2001 through 2003. There is again some similarity between the Red Willow Creek and Winters Creek EC and temperature distributions. For example, the peak temperature in 2001 is about 75° and occurs in July. The peak temperatures in 2002 and 2003 are lower than that in 2001 and in this case, are below the temperature criterion in throughout 2002 and 2003. The early 2002 and 2002-03 winter minimum temperatures in Red Willow Creek are lower than that of either of the other two previously described streams, which had winter minima in the mid-forties Fahrenheit. The early 2002 winter minimum in Red Willow Creek is in the lower thirties Fahrenheit and in the upper thirties in the winter of 2002-03. The low minimum temperatures would indicate that the influence of ground water in controlling temperature at the site is minimal during the winter.

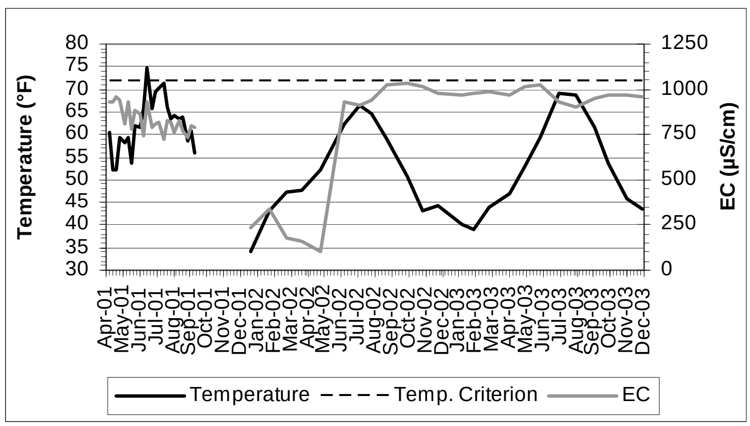


Figure 105. Red Willow Creek temperature and EC during 2001 through 2003

The EC in 2001 in Red Willow Creek shows numerous spikes, including a large one in mid-July. However, the mid-July spike reflects a return to EC levels observed in April and early May. The 2001 EC shows a general decrease from around 900 µS/cm in the spring to less than 800 µS/cm by the end of September. Unlike the other two streams, the January 2002 EC is much lower than that of the end of the preceding year, but like the other two streams, the EC decreases to a minimum in May. Also like the other two sites, the spring minimum EC is near 100 µS/cm. The winter minimum EC occurs in May like that of Tub Springs, rather than April like that of Winters Creek. Like the other two streams, the EC remains relatively constant during the summer and fall of 2002 and throughout 2003 after its increase from the spring minimum in 2002 (compare figures 102, 104, and 105).

Figure 106 shows flow and gage height data for Red Willow Creek. All of the low flows came from a site NE of Bayard, while the higher flows were from the site SE of Bayard. The low EC readings also came from the NE site. There are no gage height readings from the NE site.

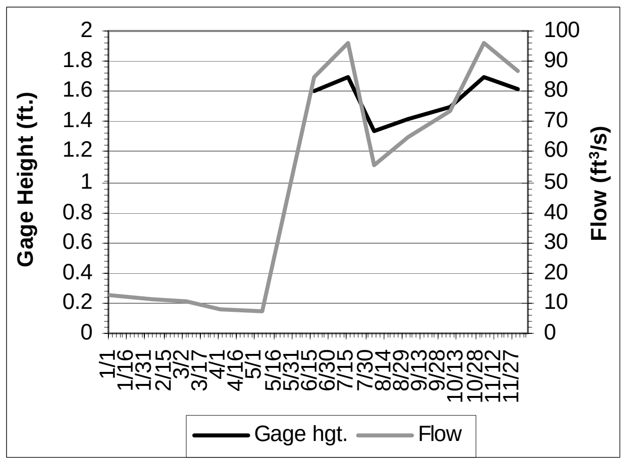


Figure 106. Red Willow Creek flow and gage height during 2002

The distribution of 2002 winter and early spring flows in Willow Creek does not differ from those of the preceding sites, despite the fact that the data are from a different site. However, the summer and fall flows differ greatly. Each of the preceding streams showed a large increase to a peak flow in August. In Red Willow Creek, there is a large decrease in flow during August (Figure 106). The August flow represents a summer minimum. Identical peak flows occur during July and November in Red Willow Creek. The increase in flow to a secondary peak is also different from what had been observed in the other two streams discussed above.

The flows in Red Willow Creek during the summer are slightly less than those in Winters Creek (compare figures 103 and 106). The associated gage height readings are also slightly less than those of Winters Creek. The data on Figure 106 indicate that the depth of water in Red Willow Creek at higher flows is intermediate between the equivalent depths of Tub Springs and Winters Creek.

Factors Affecting Flow in the Tributaries

The Sandhills cover a large area of Nebraska (Figure 107). The western edge of the Sandhills extend to near the North Platte River in Morrill County. The Sandhills intersect the north shore of Lake McConaughy and extend to the south of the North Platte River to the east of Lake McConaughy (Figure 107). A comparison with Figure 101 indicates that the Winters, Ninemile, and Red Willow creek basins drain the Sandhills, at least in part.

The Nebraska Department of Natural Resources (NDNR) operates gages on each of the tributaries listed above. There is also a gage on Tub Springs. The gage data for those streams were retrieved from the NDNR website. The periods of record (water years) include either the

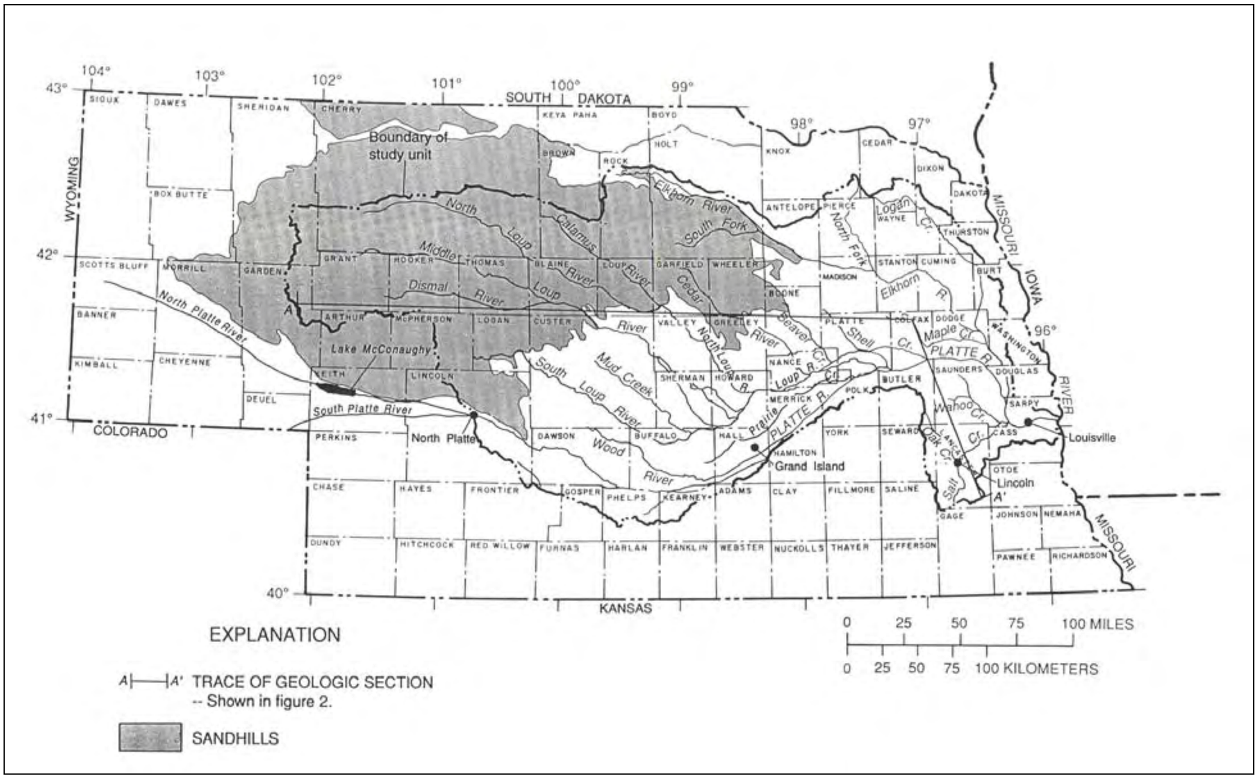


Figure 107. Geographic extent of the Sandhills in Nebraska

period 1979 through 2004 or the period 1991 through 2004. For purposes of this analysis, the period April 2001 through September 2004 are used. These flow data include the period of the NDEQ temperature and EC data described above.

The flow data for the streams are shown on Figure 108. The one thing that is obvious from the hydrographs on Figure 108 is that the flows in 2001 were much higher during 2001 than in the following three years. This would account for the EC distributions shown above for the streams for which there were data in 2002 and 2003.

Another characteristic of each of the streams is the difference between the flow distributions in the irrigations season (March or April through September or early October). During the irrigation season, the flows are extremely variable (Figure 108), while during the nonirrigation season, the hydrographs show a slow, smooth decline. The flows associated with the tributaries vary, with Ninemile Creek usually having the greatest flow and Tub Springs the lowest. There are times, particularly in 2001, when peak flows in Winters Creek exceed those of Ninemile Creek (Figure 108).

The comparison between figures 101 and 107 shows that the basins of Blue and Birdwood creeks lie almost entirely within the Sandhills. As was shown in Table 26, NDEQ also sampled both of those creeks, although in different years and only in one of the three years. The NDEQ temperature and EC data for Blue and Birdwood creeks are shown on Figure 109. In both creeks, the EC shows very little variation, with a range between 150 and 200 μS/cm. The EC of Birdwood Creek in 2001, the year in which there was the greatest degree of flow variation,

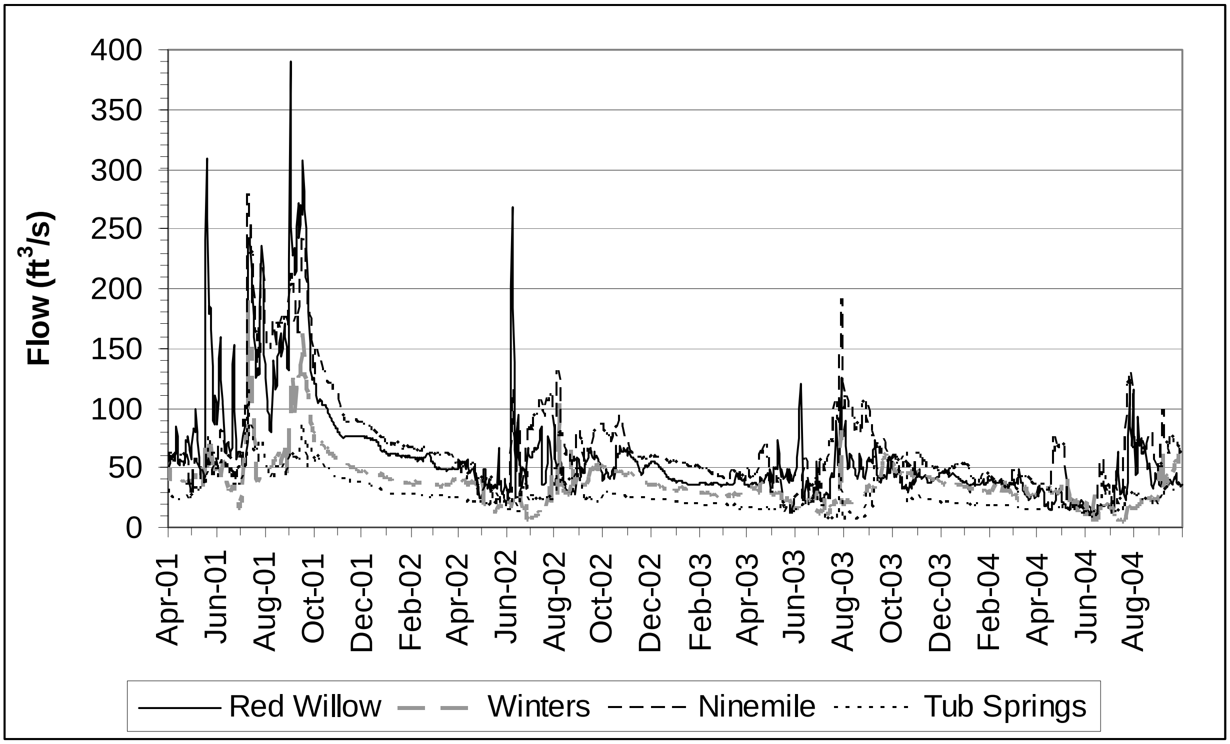
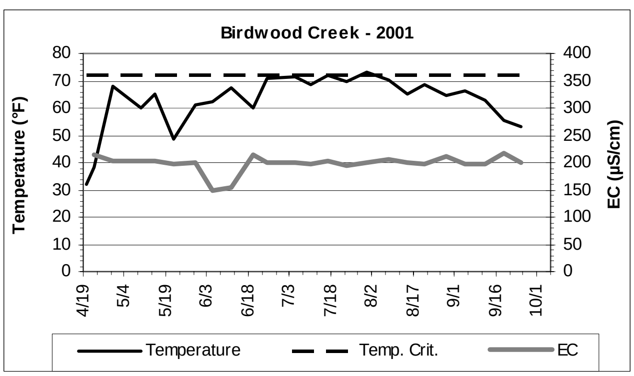


Figure 108. Hydrographs of 4 North Platte tributaries from 2001 through 2004



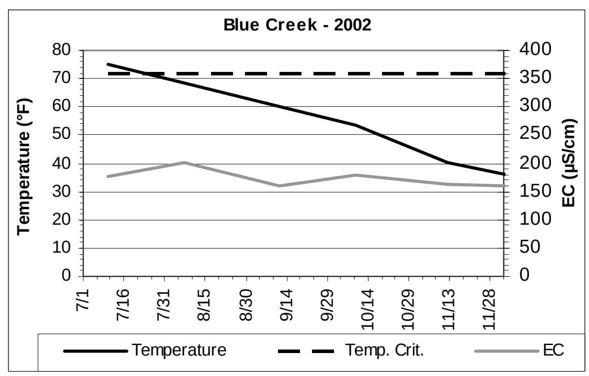


Figure 109. Temperature and EC of two North Platte River tributaries draining the Sandhills

shows a very consistent EC generally around 200 µS/cm. In contrast, the EC of Blue Creek in 2002 shows a general decrease during the autumn. The temperature in Birdwood Creek increases dramatically in late April of 2001 and remains near the temperature criterion during most of the summer. The peak temperature occurred in early August, at which time it was slightly above the temperature criterion. The data for Blue Creek only include the last part of the year, with the initial temperature measurement in early July, at which time the temperature was at its highest and somewhat above the temperature criterion at 75°F. The measurements in 2002 were made monthly and show a smooth decrease for the remainder of the year.

The EC of Blue and Birdwood creeks are on the order of the low EC described above for the four tributaries for which detailed EC data were presented. It would appear that the low EC measurements in Winters, Red Willow, and Ninemile creeks and Tub Springs that were described above are representative of the EC of the Sandhills aquifer discharge in the North Platte Basin in Nebraska. If this is the case, the Sandhills aquifer contributes infrequently to the flows in those streams and essentially only in the winter.

Figure 108 showed considerable flow variation during the irrigation season. The USGS published gage records include a remarks section that describes the factors that affect the flows above the gage. These factors are summarized in Table 28. There are factors in each of the streams that can cause flow increases and decreases. The most common factor that would increase flow is irrigation return flows. The other factor that would cause an increase in flow is spills from canals. Canal spills can be quite large and are associated with precipitation events, which could further increase stream flows. Canal spills are a likely cause of the spikes in flow that occur during the irrigation season. Alternatively, diversions for irrigation would decrease flows. The North Platte Project diverts from Winters Creek and Tub Springs. Ground water withdrawals would have a more subtle effect on flows by depleting the base flows of the streams.

Table 28. Tributaries to the North Platte River in Nebraska and factors influencing flow

Site name	Ground Water Withdrawals	Diversions for Irrigation	Irrigation Return Flows	Canal Waste (Spills)	Backwater from North Platte River
Horse Creek near Lyman	√	√	√		
Sheep Creek near Morrill	√	√	√		
Dutch Flats Drain near Mitchell		√	√		
Dry Spottedtail Creek at Mitchell	√	√	√		√
Tub Springs near Scottsbluff		√	√	√	
Winters Creek at Tri-State Canal - Scottsbluff		√	√		
Winters Creek near Scottsbluff	√	√	√		
Gering Drain near Gering			√		
Alliance Drain near Minatare		√	√		
Ninemile Drain near Minatare			√	√	
Ninemile Drain near McGrew	√	√	√		
Red Willow Creek near Bayard	√	√	√	√	√
Pumpkin Creek near Bridgeport	√	√	√		
Blue Creek near Lewellen	√	√	√		
Birdwood Creek near Hershey	√	√	√		
Lincoln Co. Drain No. 1 near North Platte			√		

Flow and Water Quality of the Interstate and Tri-State Canals

The NDNR website also has daily flow data for the Interstate and Tri-State canals. Those data were downloaded from the website. There are flow data for the Interstate Canal encompassing the inclusive water years of 1946 through 2004. The period of record for the Tri-State Canal includes the water years of 1985 through 2004. It should be noted that in the early years of the flow record of both canals the data are only for a partial year, most often beginning on April 1 and ending October 31. In some years, the entire year is included, but more often than not, the other months do not show any flow in the canals, although in some cases, the Interstate Canal began diverting water in March.

The USGS sampled the Interstate and Tri-State canals between 1995 and 1999. The samples included analyses primarily for major ions and nutrients. Temperature, DO, and EC measurements were also made in conjunction with the sample collection.

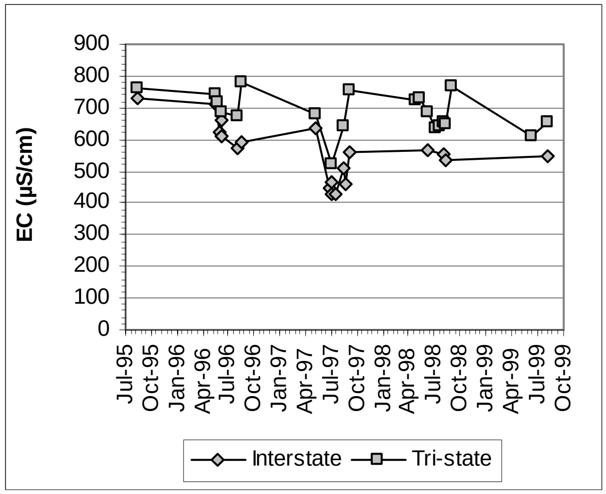


Figure 110. EC of the Interstate and Tri-State canals: 1995-99

The EC data from the two canals are plotted on Figure 110. There was only 1 sample collected from the canals in 1995 and 1 or 2 samples in 1999. The EC of the Interstate Canal is over 100 µS/cm lower than that of the Tri-State Canal on the average, *i.e.* 562 and 686 µS/cm, respectively. In both cases, both of these EC levels are well below the EC of the tributaries shown on figures 102, 104, and 105, but well above the EC of the Blue and Birdwood creeks shown on Figure 109 and assumed to represent discharge of ground water from the Sandhills Aquifer. That does not mean that neither of these sources, *i.e.* Sandhills ground water or canal water is a component of the flow in the tributaries, but it does mean that some other source is controlling the EC. The main source of higher EC water would be expected to be irrigation return flows, consisting of the subsurface discharge of deep percolation. Deep percolation is the component of returns flows that carries the salts in the irrigation water (and sometimes soil leachate) below the root zone of crops. The deep percolation carries essentially all of the salts in the subsurface water that has been depleted by evapotranspiration (ET). The EC depends on how much water was consumptively used by the crops. To raise the canal water to 900-1000 µS/cm would not require a great deal of ET. At the higher EC of the canal water, only 10 to 20 percent ET would do, while at the lower EC of the canal water, 50 to 60 percent consumptive use would effect the increase. Either of these ranges of ET would be considered low.

As was noted above, the canals do not flow year round. To illustrate this, the flows that coincide with the period of record for the EC data shown on Figure 110 are plotted on Figure 111. The flow data for the Interstate Canal from 1996 through 1998 indicate that deliveries started prior to the date that recording began. Instead of ramping up, as is usually the case, The flows were high on the first date that the data began. The main point to note on Figure 111 is that flows during most of the summer in all of the years are near 2,000 ft^3/s. The capacity of the canal is 2,200 ft^3/s; so the canal in each of the years ran near capacity throughout most or all of the later part of the irrigation season in each of the 5 years shown.

The flows in the Tri-State Canal showed various peak deliveries in the different years shown on Figure 111. The peak annual deliveries were between 1200 and 1500 ft^3/s, with the maximum occurring in 1999. The Tri-State Canal is not a Reclamation facility, but it provides water to the Northport Canal, which is an extension of the Tri-State Canal and is part of the North Platte Project. The maximum capacity of the Northport Canal is 1500 ft^3/s.

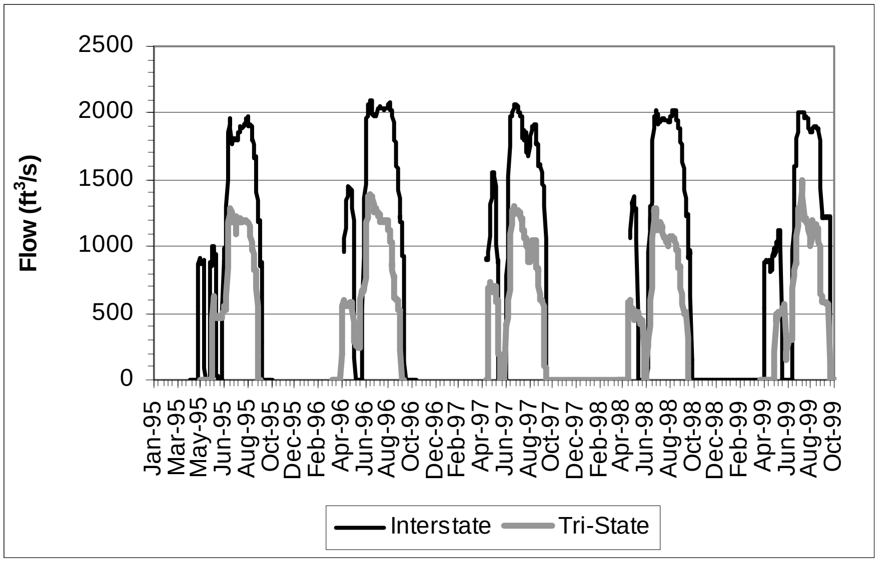


Figure 111. Flow in the Interstate and Tri-State canals
during 1995 through 1999

The Tri-State Canal diverts water from several of the North Platte tributaries between Sheep Creek and Tub Springs. These diversions could change the EC of the Tri-State Canal water as it moves away from the North Platte River. Consequently, the EC of the canal water at the farthest delivery point could be somewhat higher due to the influx of the more concentrated (higher EC) diversions from the tributaries.

The USGS temperature data from the Interstate and Tri-State canals are shown on Figure 112. Both canals show temperatures above the NDEQ temperature criterion for coldwater aquatic life in most of the years. The temperature of the water in the Interstate Canal exceeded the criterion in 3 of the 5 years for which there were temperature measurements. The water in the Tri-State Canal exceeded the temperature criterion in each of the years except 1996. Based on these data, it seems unlikely that the canal water is responsible for cooling the water in the small tributaries to the North Platte River.

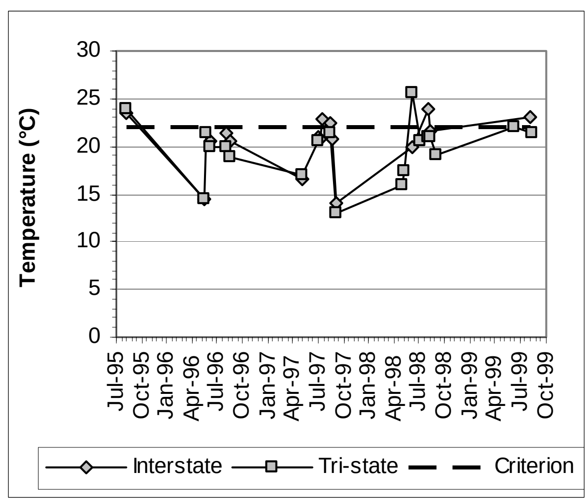


Figure 112. Temperature of the Interstate and Tri-State canals: 1995-99

The measurement sites for the canal water temperatures are made near the Wyoming-Nebraska border. With continued travel, the water in the canals would be expected to warm even more. The tributary diversions may cool the canal water slightly at times, but for the most part, the diversions are too small to have any great effect on the canal temperature. The main cooling effect in the tributaries appears more likely to be due to ground water inflow than the canal seepage.

Tub springs is used as a drain by irrigators in the area (NDNR, 2004). The mean monthly flow data for Tub Springs for the period 1985 through 2004 are shown on Figure 113. The mean monthly flow data as represented by the line labeled, gaged, indicate that the flow increases during May. The increase coincides with the usual onset of diversions into the Interstate and Tri-State canals. These results indicate that the effects of seepage are essentially immediate.

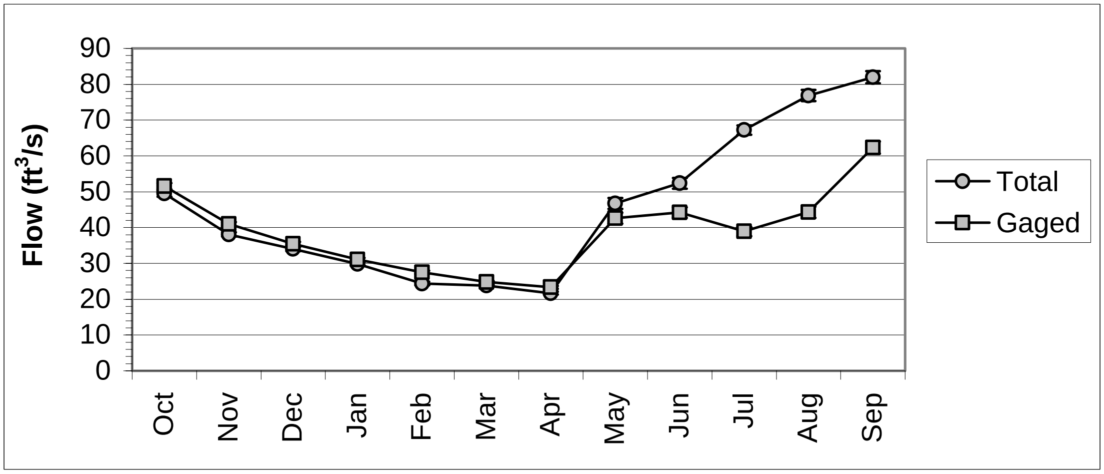


Figure 113. Flow in Tub Springs – gaged and with diversions added back (total)

The second line on Figure 113 is labeled, total. This second line represents what the flow would have been in Tub Springs at the gage if water had not been diverted into the Tri-State Canal above the gage. The total flow plot indicates that the effect of seepage at the gage would have been even more dramatic if water had not been diverted.

Figure 114 also shows a plot of the gaged data in Tub Springs, along the mean monthly flows during 2002 and 2003. In the previous section of this appendix, it was speculated that the drought conditions during 2002 and 2003 could have been responsible for the invariant EC beginning in the summer of 2002. The data plotted on Figure 114 could provide some insight into that phenomenon.

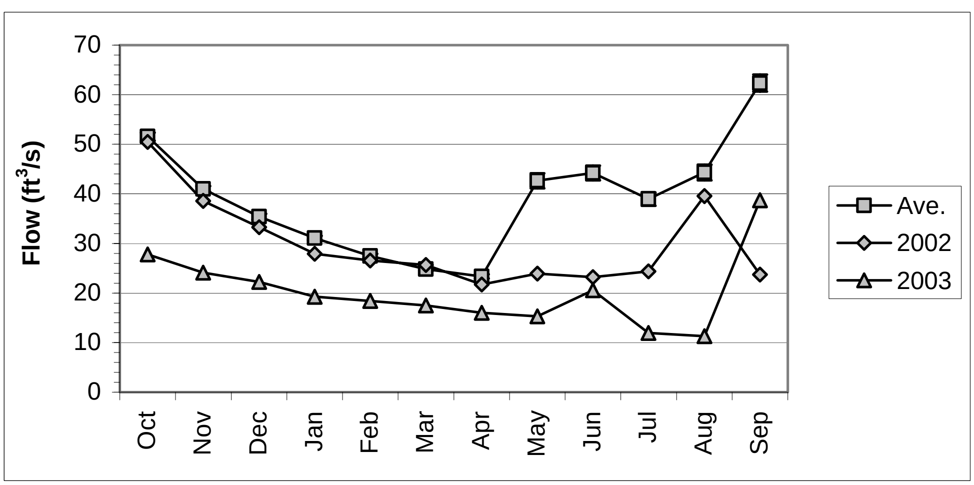


Figure 114. Tub Springs average monthly flow (1985-2004) vs. monthly averages in 2002 and 2003

The water year 2002 mean monthly flows in Tub Springs track almost perfectly along the long-term average during October through April (Figure 114). This is the period when the base flow of the stream declines and reflects a declining water table elevation. In May, the long-term average flow increases, as was noted above. However, the flow in Tub Springs during the summer of 2002 did not increase greatly until August, following which the flow once again decreased in September to approximately where it had been during July.

During water year 2003, the mean monthly flow in Tub Springs during the nonirrigation season showed the same pattern of flow as the long-term average, but the flows were between 30 and 45 percent below average. There was a small increase in flow during June, but the flows dropped to their lowest of the year during July and August. Recovery began to appear in September and continued into October. However, after October, the mean monthly flows in 2004 were virtually identical to those of 2003.

A further indicator of the effects of the drought in the area of the Panhandle irrigation districts is illustrated on Figure 115, which shows the daily flows in the Interstate and Tri-State canals during the most recent water years for which data are available. The pattern of canal flows in 2001 is similar to what was shown for the 1995 through 1999 period on Figure 111.

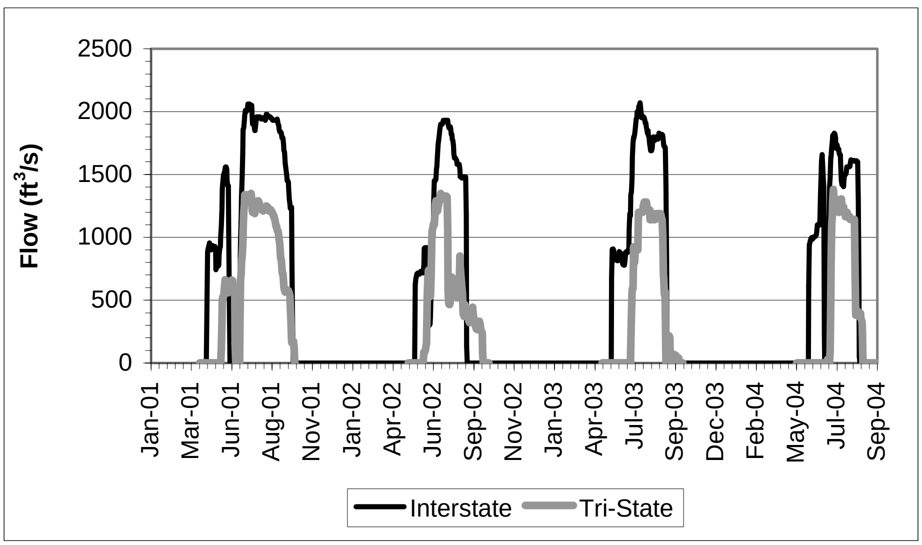


Figure 115. Flow in the Interstate and Tri-State canals during 2001 through 2004 (period of NDEQ data from tributaries and most recent water year)

In the Interstate Canal, there were early deliveries of 1,000 to 1,500 ft^3/s in March and April of 2001 (Figure 115), as was the case in the 1995 through 1999 period. These were followed by deliveries of around 2,000 ft^3/s during June through August, with deliveries decreasing in September. The pattern of flows during 2002 through 2004 was rather different from 2001. The spring peak delivery in 2002-04 still occurred, but the maximum flow was less than 1,000 ft^3/s. Although the peak deliveries in 2002-04 were still in the 2,000 ft^3/s range, their duration was much shorter, and deliveries ended in August.

The deliveries in the Tri-State Canal like those in the Interstate Canal showed a decline during 2002-04, although the annual pattern in 2002 also differed somewhat from those of 2003 and 2004. In 2002, the Tri-State Canal deliveries did not show the spring peak that had been evident in previous years figures 111 and 115). Deliveries in 2002, unlike most years, dropped off in July, rather than in September. Alternatively, in 2003 and 2004, deliveries of water in the Tri-State Canal did not begin until July, then continued into August (Figure 115). Obviously, the drought condition affected deliveries to the participating irrigation districts in the North Platte Project.

Water supply conditions through the North Platte system were below average in water year 2004 (Reclamation, 2005). The inflow to Seminoe Reservoir was only 38 percent of average. The maximum storage in Pathfinder Reservoir, which serves the Interstate Canal, was only 33 percent of capacity. A summary of the water supply conditions from Reclamation (2005) states:

> For the third year in a row because of low carryover storage, drought conditions and below average snowmelt runoff, an allocation of storage water was put into effect on June 16, 2004. The allocation applied to the four Government Districts, (Pathfinder Irrigation District (ID), Goshen ID, Gering-Fort Laramie ID and Northport ID) and to the nine Warren Act Contractors, (Farmers ID, Gering ID, Lingle Water Users Assoc., Hill ID, Rock Ranch ID, Central ID, Chimney Rock ID, Browns Creek ID, and Beerline Irrigation Canal Co).

The above indicates that water supply in the North Platte Basin was below average during the period 2002-2004. This would indicate that the data on the tributaries from 2002 and 2003 are not representative of long-term conditions. The data from 2001 may be representative, but those data only encompass a part of the year, but that part of the year is critical for cold water fish in terms of temperature limitation.

Interrelationships among Sites

Table 27 showed correlations among variables from Winters Creek and Tub Springs. The sites are not physically connected. The correlations indicate that there are similarities among the controls on temperature and flow in the two streams. Table 29 shows a similar set of correlations between the temperature and flow data among all of the sites that were sampled in 2002. Once again, there is no physical control of one site over another. As was noted above, temperature is primarily controlled by insolation, which should be similar across a region, such as the North Platte Basin in Nebraska. Table 29 indicates that this is so. The temperature data from all of the sites are very highly correlated, including the temperature of the sites on the North Platte River mainstem.

The mainstem sites are located upstream and downstream from Lake McConaughy. The temperature of the downstream site at North Platte (NP-NP in Table 29) should be at least partially controlled by the reservoir. Given the vary high correlation coefficient (r-value) between the temperatures at the two sites, the reservoir does not appear to exert that much control. Recall from Figure 15 the offset between the monthly outflow temperature of Seminoe Reservoir and that of the free-flowing Sweetwater River. This is not evident in the two North Platte sites near Lake McConaughy. Usually, the effect of a reservoir does not extend more than 10 to 20 miles downstream, unless the river is located in a deeply entrenched canyon. As long as the river is exposed to solar radiation, the typical pattern will return.

Table 29. Correlations of temperature and flow at 2 sites on the North Platte River and sites on selected North Platte tributaries					
2002 Temperature correlations – r-values					
	NP-NP	*NP Lewellen*	*Ninemile*	*Red Willow*	*Tub Springs*
NP Lewellen	0.9837				
Ninemile Cr.	0.8733	0.8616			
Red Willow	0.9273	0.9144	0.9429		
Tub Springs	0.9546	0.9492	0.8984	0.9468	
Winters Cr.	0.9108	0.9121	0.9404	0.9767	0.9536
All with probability of a greater r < 0.001					
2002 flow data – r-values					
	NP-NP	*NP Lewellen*	*Ninemile*	*Red Willow*	*Tub Springs*
NP Lewellen	-0.2426				
Ninemile Cr.	**0.6869**	-0.5343			
Red Willow	0.3256	-0.5201	**0.6830**		
Tub Springs	0.3823	-0.0547	**0.6078**	-0.0285	
Winters Cr.	0.2394	-0.0281	0.5479	-0.0215	**0.8304**
Bold: Prob. > r > 0.01 & < 0.05 = minimally significant					

The reason for comparing the data from the sites is to evaluate how much effect ground water might have on the temperature of the various sites. It was noted above, that there was an apparent control on the TDS of the North Platte River at Lisco/Lewellen. Based on the preceding relationship between the temperatures of the two North Platte River sites, this control does not extend to temperature. Ground water can exert a control on temperature similar to that of a reservoir. For this analysis, the working hypothesis is that the sites that have the poorest correlations with the temperature of the North Platte River will be those with the greatest ground water influence.

The r-values of the correlations between the temperatures of each of the tributaries with those of the North Platte River sites and among the temperatures of the tributaries are all greater than 0.9, with the exception of several for Ninemile Creek (or Drain). The poorest correlations (lowest r-values) are between the temperature of Ninemile Creek and the two North Platte River sites. The next lowest r-value is from the correlation between the temperatures of Ninemile Creek and Tub Springs, although the r-value for that correlation does round to 0.9. The temperature of Tub Springs shows the best correlation with the North Platte River sites of any of the tributary sites.

The lower part of Table 29 shows correlations among the monthly flows at the various sites. Of most interest is the negative r-value between the flow at the two North Platte mainstem sites. The fact that the r-value for the correlation between flows at the two North Platte River sites is negative shows the effect of operations of the North Platte Project on the Lewellen gage and CNPPIDs operation of Lake McConaughy (actually, Keystone Diversion) yield much different flow regimes at the two sites.

The most significant correlation among flows is the one discussed above between Tub Springs and Winters Creek. This is followed by correlations of the flow of Ninemile Creek with those of the North Platte at North Platte, Red Willow Creek, and Tub Springs, respectively. None of the other flow correlations are statistically significant.

All of the low flow data for Ninemile Creek were collected from the Minatare site during 2002. The relationship of flow between the 2 sites that was derived above was applied to these flows to estimate the flow at the McGrew site. The correlations between the flow of Ninemile Creek and the other sites shown in Table 29 were recalculated. The revised correlations are shown in Table 30. The most dramatic changes in the flow correlations based on the adjusted Ninemile Creek flows are in those with the Tub Springs and Red Willow Creek flows. The former correlation between the flows of Ninemile and Red Willow creeks is no longer significant, while the correlation between flows in Ninemile Creek and Tub Springs is much increased and is now highly significant, as is the correlation between the Ninemile Creek and North Platte at North Platte flows. There is also now a significant correlation between the Ninemile Creek and Winters Creek flows. Meanwhile, the correlation between the flows of Ninemile Creek and the North Platte at Lewellen is diminished.

Table 30. 2002 flow data with adjusted Ninemile flows

	Adjusted Ninemile
NP - NP	**0.7584**
NP - Lewellen	-0.2511
Red Willow	0.2691
Tub Springs	**0.7983**
Winters Creek	**0.6879**
P < 0.05	
P < 0.01	

Recall that the low flow data from both Ninemile Creek and Red Willow Creek were collected from sites different from the high flow data. The Ninemile Creek data can be adjusted using the earlier derived regression relationship, while the Red Willow Creek data cannot. The adjustment of the Ninemile Creek flow data changes its relationship to other sites rather dramatically. This raises the question concerning the other low-flow data from the other sites and how those data affect the Red Willow Creek results described above. It seems likely that the low-flows should be higher. However, the possible related differences in the low-flow temperature and EC data cannot be defined.

Drains

The Gering Drain was only sampled during 2001. Because of the complications with the 2002 data and to be consistent with the Gering Drain data, this section will include only the 2001 data from the Ninemile Drain.

Figure 116 shows the temperature and EC of the Gering Drain during the 2001 irrigation season. The EC of the Gering Drain showed a rather dramatic decrease from a high of about 1200 µS/cm in mid-May to 700 µS/cm in mid-June. Meanwhile, the temperature was above the criterion from late June through the middle of August. The peak temperature in 2001 was over

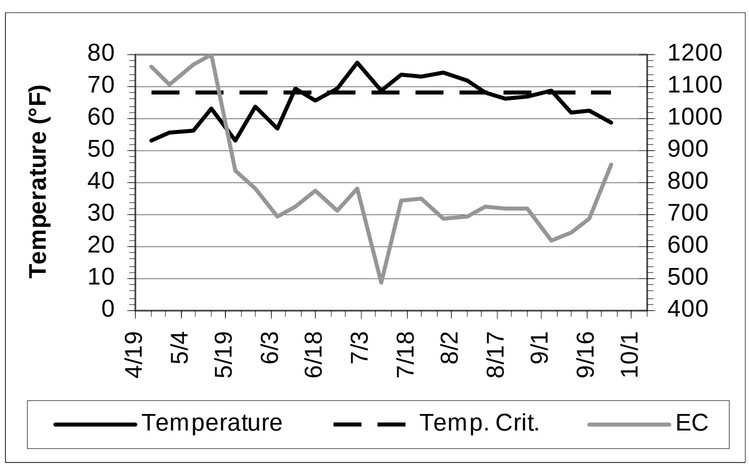


Figure 116. Gering Drain temperature and EC during the 2001 irrigation season

77° and nearly 40 percent of the samples exceeded the temperature criterion. The Gering drain is not listed among the Panhandle trout fisheries by NG&PC (2005), and based on the 2001 data, it does not seem likely that it could support a trout fishery. The drain data would indicate that the drainage from irrigation is actually warmer than what was shown in the tributaries described above.

The very large decrease in EC in July (on the 10th) on Figure 116 appeared to be due to the influence of storm runoff. There are no flow data for the 2001 samples. However, there are data on total phosphorus (TP) and total suspended solids (TSS). Both of these tend to be heavily influenced by erosion during storms. Figure 117 shows the concentrations of TP and TSS in the Gering Drain during 2001. As is evident, there is an extremely large increase in both TP and TSS that coincide with the date of the low EC. Similar large increases in TSS occurred in most of the other tributaries and in the North Platte River as well, indicating a rather widespread storm occurred on July 10th. Based on this evidence, it seems safe to conclude that there was a significant storm on July 10th that contributed a large inflow of dilute water to the Gering Drain, among others. This also coincided with a decrease in temperature in the Gering drain, marking the only sample date in July 2001 that the temperature was below 72°. Nevertheless, the temperature on July 10th was still slightly above the temperature criterion for trout in the Gering Drain.

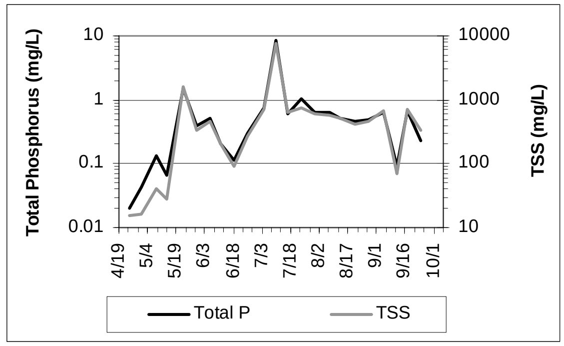


Figure 117. Gering Drain total phosphorus and total suspended solids in 2001

It is rather obvious on Figure 118 that TP and TSS track very well during 2001 in the Gering Drain. This is so much so that the correlation coefficient between the TP and TSS data on Figure 118 is 0.998. This is the type of relationship that was sought earlier in the North Platte River in Wyoming, but not found.

Ninemile Creek (or Drain) is listed as supporting brown and rainbow trout in NG&PC (2005). The weekly temperature and EC data from Ninemile Creek in 2001 are shown on Figure 118. Although not as dramatic as in the Gering Drain, there is also a decrease in EC during the summer of 2001. Where the EC in the Gering Drain was initially over 1100 µS/cm, the initial EC in Ninemile Creek was between 900 and 1000 µS/cm. Both decreased to between 700 and 800 µS/cm.

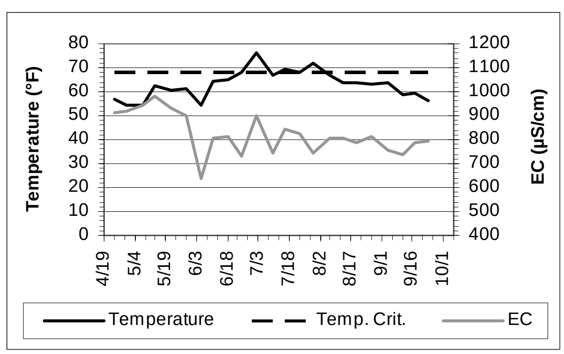


Figure 118. Ninemile Drain temperature and EC in 2001

There was also a decrease in EC on July 10 in the Ninemile Creek. However, the EC on that date was not lower than the EC on several other dates. The minimum EC occurred on June 3 with a reading of 630 µS/cm. There was no indication of a large source of dilution water in Ninemile

Creek on June 3rd, but this was the low point of the decrease in EC that began in May. The EC remained centered on 800 µS/cm for the remainder of the monitoring period (Figure 118).

The peak temperature in Ninemile Creek during 2001 was 76°, which occurred on the same date as the peak in the Gering Drain. However, where the Gering Drain temperature remained high through most of July, the temperature in Ninemile Creek did not exceed 72° during the remainder of the month (Figure 118).

USGS Ground Water Studies in the Nebraska Panhandle

The USGS has conducted a number of studies of ground water in the Dutch Flats area. In an early study, Babcock and Visher (1951) observed that the water table rose from June to September and declined during the intervening months. Although the Dutch Flats area includes North Platte tributaries between Sheep Creek to the west and Spotted Tail Creek to the east only, the pattern is consistent with the discharge pattern shown on Figure 113. Babcock and Visher (1951) also indicated that between 67 and 74 percent of the water that is diverted from the North Platte River between Whalen Diversion Dam and Bridgeport returns to the river, an area that would also include Tub Springs. They also indicate that most of the ground water discharged from the ground-water reservoir in the Dutch Flats area is primarily thorough streams and drains.

Verstraeten *et al.* (2001) surface water-ground water interactions in the Dutch Flats area. Their results indicate that seepage from the Interstate Canal controlled the ground water in its vicinity. At the Interstate Canal, surface water seeping from the canal appeared to replace ground water in about 1 month in the upper 30 feet of the aquifer within about 1 mile of the canal. One result of this study that was pertinent to the above EC analysis was that the EC of the ground water increased when water was flowing in the Interstate Canal, but they indicate that a range of 774 to 896 J,.lS/cm is typical of alluvial ground water of Sheep Creek and Dry Sheep Creek. This would indicate that there is some support for using EC as an indicator of a ground water influence in the North Platte tributary streams.

Verstraeten *et al.* (1995) conducted a reconnaissance study of ground water quality in the North Platte Natural Resources District in the summer of 1991. Temperature measurements were made in 74 wells in the unconfined Quaternary aquifer. The temperature ranged from 9 to 18°C (48-64°F). The equivalent to the upper 95 percent confidence interval was 16°C (61°F). The coincidental range in EC was from 329 to 1,460 µS/cm. Although these results do not show anything definitive in regard to EC, it does show that the temperature of the ground water is low enough to meet the coldwater aquatic life criterion. Because the peak water temperatures in the Interstate and Tri-State canals exceed the coldwater aquatic life criterion**,** it seems highly probable that the ground water mass, which is large enough to buffer against warming from canal seepage and irrigation deep percolation, is the control on temperature in the tributaries.

The preceding summaries involved tributaries on the north side of the North Platte Valley. Verstraeten *et al.* (2001) also provided an overview of Horse Creek and its associated ground water. The flows in Horse Creek ranged from 20 to about 700 ft^3/s during 1995 through the end of water year 1998. On the basis of the North Platte River and Horse Creek hydrographs, Verstraeten *et al.* (2001) conclude that flows at the natural surface-water sites generally increased near the beginning of May and began to decline sometime in August. The hydrograph for Horse Creek generally resembles those presented on figures 91 through 93 above. The Mitchell Canal crosses Horse Creek above the gage used to generate the hydrographs, but there is no discussion of seepage associated with the Mitchell Canal.

Another area to the south of the river that has been studied is the Pumpkin Creek valley. A recent study by Steele *et al.* indicates that ground water levels in the valley have been declining for the last 10 years. NDNR (2004) shows that there was no flow in the creek during much of the year, but a plot of the data (Figure 119) retrieved from the NDNR website indicates that the loss of base flow is more a reflection of the drought than of the effect of any long-term trend related to ground water levels, at least at this point.

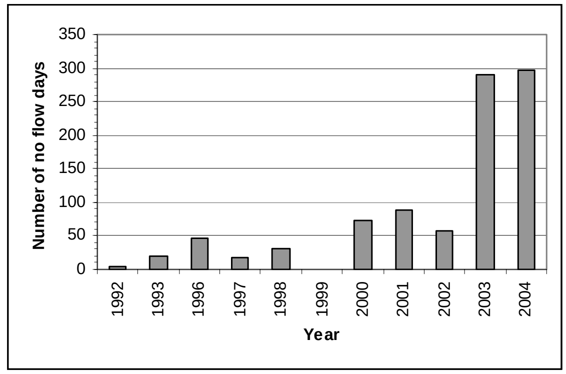


Figure 119. Number of no flow days per year in Pumpkin Creek

To put things into perspective, according to NDNR (2004), there are three diversions upstream from the Pumpkin Creek gage. Two of the three canals diverted no water during 2004. Alternatively, the gage is located downstream from the Belmont Canal, which diverted water from mid-April through the end of September of 2004. Apparently, under the conditions that existed in 2004, canal seepage did not have great effect on the Pumpkin Creek flows at the gage.

Temperature Summary

The above analysis is based on data from specific sites, many of which are located at the lower end of tributaries. These locations may be relatively far from ground water influences. Although the temperatures at the sites described above may not appear suitable for trout, at least at times, there are likely areas in the streams with temperatures suitable for trout, *i.e.* refugia, where the trout can survive until suitable temperatures return to the greater part of these streams. In addition, coldwater fish can survive unsuitable temperatures for brief periods, although the fish would undergo stress. Stress in itself has an impact. Most common among the sublethal effect of stress relates to feeding. Feeding and thus growth ceases and weight loss could occur. If these effects should occur during gametogenesis, reproduction could fail and a year-class of fish would be lost.

Most of the water quality data used in the above were collected in 2001 through 2003. All of the three years are drought years – 2002 was the most severe drought year of record and 2003 ranked second for deliveries to the Interstate Canal, based on the period 1946 though 2004. This calls into question how representative the above data are. For example, the EC in the tributaries showed a decrease in 2001, but dramatic increases in the summer of 2002 and no decrease throughout 2003. The results shown may be more of an exception than the rule. If so, they are probably representative of the types of conditions that would be as bad or possibly worse than anything that could be caused by the Program alternatives.

Dissolved Oxygen

Coldwater fish tend to be rather intolerant of low DO. In the above referenced Lake McConaughy study, a DO criterion of 3 mg/L was used for the Lake McConaughy rainbow trout, However, most trout species (or strains) require much higher DO than that, more in the range of 5 to 7 mg/L. The Nebraska coldwater aquatic life DO criteria are a daily minimum of 8 mg/L of salmonid early life stages are present and 4 mg/L if early life stages are absent. Because of the potential sensitivity of trout to low DO, the NDEQ data were explored for low DO concentrations. Table 31 summarizes the minimum DO concentrations in each of the three streams discussed above.

Table 31. Minimum dissolved oxygen in four North Platte River tributaries

Stream	2001	2002	2003
Winters Creek	7.6	4.0	6.7
Tub Springs	—	4.2	4.9
Red Willow	7.5	4.2	7.1
Ninemile Cr.	7.8	5.0	7.4

The minimum DO during 2001 was above 7 mg/L in Winters, Red Willow, and Ninemile creeks. Tub Springs was not sampled during 2001. The minimum DO in 2002 was around 4 mg/L in all of the streams, except Ninemile Creek, where the minimum DO was 5 mg/L. Coincidentally, the low DO in each of the streams in 2002 occurred during April. Because of this coincidence the original data file was further reviewed. There was nothing in the remarks column to indicate a data quality problem. The data were sorted by date and sampler to see if maybe there was an indication of an instrument problem. Although all of the low DO readings were made in the afternoon of April 2, 2002, they showed no consistent pattern. There were higher values, *i.e.* greater than 5 mg/L, recorded between some of the readings shown in Table 31. During 2003, the minimum DO in all of the streams increased to around 7 mg/L with the exception of Tub Springs, where the minimum was once again below 5 mg/L. The minimum DO during 2003 occurred in May in three of the four tributaries; in the fourth tributary (Red Willow Creek), the minimum DO concentration occurred in August, but the concentration was less than 0.2 mg/L less than what had been observed in May. The minimum, or at least low, DO concentrations appear to be somewhat seasonal and occur in the spring in the tributaries shown in Table 31. If the streams are used for spawning by trout, the emergent fry should be present during the spring months when the DO minima occur. In this case, the DO criterion would be 8 mg/L and none of the minimum DO concentrations would meet it. Alternatively, if there were no spawning in the streams, the DO criterion would be 4 mg/L, which is exceeded by all of the DO minima (Table 31).

All of the low DO readings in 2002 were associated with low EC readings. The low EC readings were interpreted as signifying native ground water. It is not unusual for ground water to show seasonally low DO. There were no low EC readings during 2003; however, the minimum DO readings were not as low during 2003 either. As a consequence, the interrelationship of ground water inflows on DO in the tributaries cannot be ruled out. The low DO association with low EC water implies that the ground water influence does not originate from canal seepage, but rather from native ground water with a linger residence time in the aquifer. The longer residence time would allow for greater depletion during travel between the recharge area and the discharge point in the tributaries. The 2003 data would indicate that all of the base flow in the tributaries originated from canal seepage.

Effects of the Water Leasing Alternative

The Water Leasing Alternative would include leases in the North Platte Project irrigation districts in the Nebraska Panhandle. Although there is no way to tell at this point exactly where the leases would occur, it is likely that some would be in the area serviced by the Interstate Canal, as well as the Fort Laramie and Northport canals. Diversions from the North Platte River, which have been bypassed at the Whalen Diversion Dam, are conveyed to the Northport Canal through the Tri-State Canal. Consequently, water leases have the potential to decrease deliveries in the Interstate, Fort Laramie, Tri-State, and Northport canals.

It should be noted that the water leasing in the Nebraska Panhandle would actually be implemented is the subject of considerable debate. Water operations in the North Platte Basin are subject to agreements and laws governing water rights. The use and quantity of water are allocated for certain defined purposes - some on a priority basis, some on a proportionate share basis, and some on a geographical source basis (Reclamation, 2006). However, for purposes of evaluating the effects of the Full Water Leasing Alternative, it will be assumed that leasing can be implemented.

The decrease in deliveries would decrease seepage from canal, laterals, and on-farm irrigation applications. The decrease in seepage would decrease ground water recharge in the affected area. Based on comments on the DEIS, the tributaries that lie between the Interstate Canal and the North Platte River are of most concern related to the effects of the Full Water Leasing Alternative. Verstraeten *et al.* (2001) summarized the conditions between Sheep Creek and Dry Spottedtail Creek as follows:

> Tributaries of the North Platte River, which are much smaller in terms of discharge than the Interstate or Tri-State Canals, drain much of the study area north of the North Platte River. The largest tributary north of the river, Sheep Creek, is perennial only in the downstream reaches. In the upstream reaches, Sheep Creek typically flows only when ground water in the area is recharged with seepage from the Interstate Canal. Seepage from this canal causes ground-water levels to rise, resulting in discharge of ground water to the upstream reaches of Sheep Creek. Other tributaries of the North Platte River that lie north of the river are Dry Sheep Creek, which is perennial, and Spottedtail and Dry Spottedtail Creeks (fig. 1), which are seasonal.

Based on this summary, a reduction in flow in the Interstate Canal would reduce the flow in the listed streams.

Babcock and Visher (1951) attempted to estimate the seepage and resulting ground water recharge during 1949 in the Pathfinder Irrigation District, which is served by the Interstate Canal in the area between Dry Sheep Creek and Dry Spottedtail Creek. Seepage and recharge were estimated by two methods, which served to bracket the actual values. The first method, based on specific yield applied to changes in the annual hydrograph in wells, gave an estimate of about

53,200 acre-feet per year over the area between the creeks (about 20,000 acres). The second method, based on a water balance between the diversions and deliveries to laterals and farms, gave an estimate of 59,000 acre-feet per year. Babcock and Visher (1951) indicated a best estimate was probably somewhere between the two estimates and was most likely around 56,000 acre-feet per year.

As was shown on figures 111 and 115, deliveries by the Interstate Canal varies depending on supply conditions in the North Platte Basin. It was noted above that the years 2001-2003 were drought years. To investigate this further, the complete records for a selection of tributaries, including those shown above on Figure 108, was downloaded from the NDNR website, along with the records for the Interstate, Tri-State, and Northport canals. Annual flows in acre-feet were calculated. Annual flows were used to minimize the effects of travel time in the tributaries. Correlations among the annual flow records for the tributaries and the canals using the period 1985 through 2004 are shown in Table 32. Correlations between the tributary flows and year were also calculated to see if there was any trend over time. These are also shown in Table 32.

Table 32. Pearson correlation matrix – annual flows

Variable		Pumpkin	Red Willow	Winter Cr.	Ninemile	Tub Springs	Gering Dr
Year	r	-0.8841	-0.4494	-0.4606	-0.6793	-0.4964	-0.2844
	Prob. > r	0.000060	0.046826	0.040994	0.000988	0.030631	0.346250
	n	13	20	20	20	19	13
Interstate	r	0.5943	0.7870	0.8126	0.7618	0.6573	0.8752
	Prob. > r	0.032215	0.000038	0.000013	0.000095	0.002227	0.000089
	n	13	20	20	20	19	13
Tri-State	r	0.6163	0.8319	0.8389	0.8013	0.7438	0.8914
	Prob. > r	0.024898	0.000005	0.000004	0.000022	0.000262	0.000043
	n	13	20	20	20	19	13
Northport	r	0.6466	0.8648	0.8780	0.8712	0.7874	0.9303
	Prob. > r	0.023069	0.000002	0.000001	0.000001	0.000105	0.000011
	n	12	19	19	19	18	12

The "n" shown in Table 32 represents the number of years in the record used in the correlations. The full record of 1985 through 2004 would total 20 years. Some of the tributaries had records beginning later than 1985; so their records include less than 20 years. In the case of the Northport Canal, there was only a partial record for 1998, which gave a total annual flow of 9,000 acre-feet; because of the artificially low total flow, the data for 1998 for the Northport Canal were discarded.

As was noted above, Pumpkin Creek is a south bank tributary to the North Platte River. As such, its flow cannot be affected by any of the 3 canals shown in Table 32. Steele *et al.* (2005) indicate that there are sections of Pumpkin Creek that receive ground water discharge. Steele *et al.* (2005) also indicate that the ground water in the area has been declining. The ground water decline was one of the reasons for Steele *et al.* (2005) undertaking the study in the Pumpkin Creek valley.

Steele *et al.* (2005) attempted to estimate the age of ground water in the Pumpkin Creek valley. One of the sites with the youngest ground water, *i.e.* less than 10 years, was located in a shallow well near the mouth of Pumpkin Creek just downstream from the Belmont Canal [see Steele *et al.* (2005), Figure 96a]. The Belmont Canal diverts from the North Platte River (NDNR, 2004). Both the well and the canal crossing are located upstream from the Pumpkin Creek gage. The canal flows may be similar enough to those of the three canals shown in Table 32 to account for their slightly significant correlation with the Pumpkin Creek flows.

The annual flow in Pumpkin Creek also shows a highly significant inverse correlation with year (Table 32), based on the probability shown on the second line of each of the correlation output summaries (*i.e.* 6 in 10,000). The above-referenced decline in the water table in the area would lead to a decrease in ground water gains in the stream. This decrease would lead to a decrease in flow in the creek and could account for its inverse correlation with year.

With the exception of the Gering Drain, each of the other tributaries also show a significant inverse correlation with year, although none of these is quite as significant as that for Pumpkin Creek (Table 32). These slightly significant correlations with year may reflect the drought conditions at the end of what are relatively short periods of record. Alternatively, the correlation for Ninemile Creek is significant enough that it may indicate a longer term trend than the drought would indicate.

All of the north bank tributaries to the North Platte River show very highly significant correlations between their annual flows and those of the canals. In each case, the most significant correlation for the tributaries is with the flow in the Northport Canal, which is both the smallest and the easternmost of the 3 canals. Nevertheless, the correlations between canal flows, which reflect a given year's water delivery, and the flows in the tributaries does indicate that there is a significant effect due to the influence of the canals, whether it be due to canal seepage, lateral seepage, or surface and subsurface irrigation return flows.

The question remains concerning the influence of those flows on water temperature in the tributaries. Table 33 shows correlations between temperature and flow in each of the tributaries. Because temperature data for 2 of the 3 years consist of monthly values, the data from 2001, which were weekly, were averaged by month. There is no significant relationship between temperature and either flow or its log-transformation. In the tributaries with the most data (greatest "n"), the r-values between flow and temperature are positive, indicating an increase in temperature with an increase in flow. Because the relationships are not significant, the results in Table 33 would indicate that water leasing could at best have only a minimal impact on temperature in the tributaries.

Table 33. Correlations between temperature and flow in 5 North Platte Tributaries

Variable		Pumpkin	Red Willow	Winter	Ninemile	Tub Springs
Flow	r	-0.2167	0.2356	0.0271	0.3623	-0.0167
	Prob. > r	0.345521	0.218645	0.889021	0.053429	0.938422
	n	21	29	29	29	24
Log Flow	r	-0.1437	0.2404	-0.1830	0.2912	-0.2023
	Prob. > r	0.534382	0.209154	0.341966	0.125362	0.343167
	n	21	29	29	29	24

Ideally, to define the potential impacts of changes in flow on water temperature, a longer period of record would be desirable. A better analysis would be gained by using annual flows or summer-time flows for correlations with the maximum temperature for the year. In the absence of such a data set, the preceding analysis is about the best that can be accomplished.

Inland Lakes

The Inland Lakes, the largest of which Lake Minatare, are part of the North Platte Project. The Inland Lakes are offstream reservoirs located near Scottsbluff (Figure 120). The lakes are filled via the Interstate Canal from diversions at Whalen Dam. Lake Minatare will be the focus of this section of the appendix, not only because it is the largest, but also because there are recent water quality data available on the reservoir.

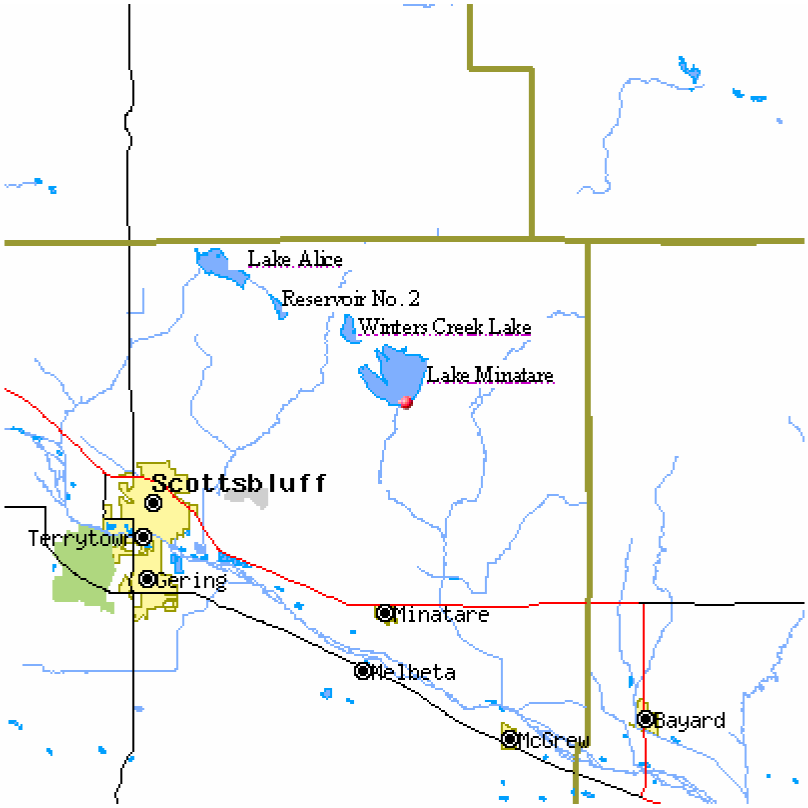


Figure 120. Location map of the Inland Lakes

The NDEQ sampled Lake Minatare during 2004. A deep-water site was sampled 3 times (May, July, and August). Temperature, DO, pH, and EC profiles were measured at 2 sites, the deep-water site and a mid-lake site during May and July. Surface samples were also collected from the deep-water site on each of the sample runs. Secchi depths were measured with the profiles at the deep-water site. Surface samples were analyzed for chlorophyll, nutrients, turbidity, and total suspended solids.

Figure 121 shows plots of the temperature and DO profiles from the deep-water site for the 3 months in which they were measured. The profiles from the mid-lake site are similar to those from the deep-water site and are not presented.

The profiles on Figure 121 are typical of Nebraska warm-water Plains reservoirs. There is little evidence of thermal stratification. The May and July profiles are relatively well-oxygenated throughout, although there is some evidence of DO depression in the deeper waters. The absence of stratification allows the reservoir to mix throughout the length of the profile. This is considerably different from the North Platte mainstem reservoirs is Wyoming, where stratification occurs and a high degree of late summer DO depletion at depth is common. In an unstratified reservoir, mixing conveys atmospheric oxygen to the depths of the reservoir in sufficient quantities to replenish most of the DO used by the sediments during organic decomposition. However, There is some degree of DO depression throughout the August profile in Lake Minatare, indicating that oxygen demand exceeds the aeration rate and is probably not entirely due to sediment oxygen demand.

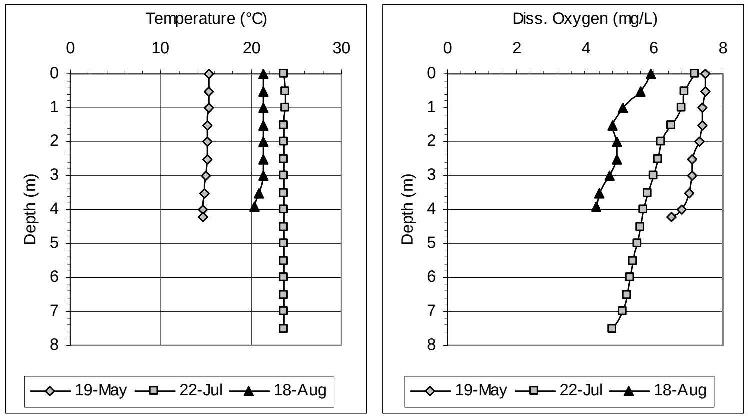


Figure 121. Temperature and DO profiles from the Lake Minatare deep-water Site during 2004

According to NG&PC (2005), Lake Minatare supports a fishery consisting of channel catfish, crappie, smallmouth bass, white bass, walleye, wiper (white and striped bass hybrid), and yellow perch. Since 1998, the lake has been actively managed to promote walleye reproduction with the construction of spawning beds (Grier, 2005). Forage fish (gizzard shad) are stocked annually to provide a food base to promote growth (*ibid.*). The temperature and DO are suitable for supporting such a warm water fishery.

The analytical results from the surface water samples from the deep-water site are shown in Table 34. The chlorophyll concentrations are indicative of mild phytoplankton bloom conditions in the spring and fall. The slightly elevated chlorophyll concentrations in the spring typically reflect a bloom of diatoms. The elevated late summer chlorophyll concentrations are likely due to a mild bloom of the Cyanobacterium, *Aphanozomenon flos-aquae*, a phenomenon that occurs in late-July to early August throughout the North Platte mainstem reservoir system from Seminoe Reservoir to Lake McConaughy. The chlorophyll concentrations in late summer in the mainstem reservoirs are much higher than those shown in Table 34 during August.

Table 34. NDEQ Surface samples from Lake Minatare during 2004

Constituent	Units	5/19	7/22	8/18
Chlorophyll	mg/m^3	5.22	3.37	6.65
Conductivity	µmhos/cm	745	814	796
Dissolved Ortho-Phosphate	mg/L	< 0.02	< 0.02	< 0.02
Total Phosphorus	mg/L	0.11	0.05	0.06
Nitrate + Nitrite, as N	mg/L	< 0.05	< 0.05	< 0.05
Total Kjeldahl Nitrogen	mg/L	0.85	< 0.5	0.68
pH-Field	S.U.	8.3	8.3	7.4
Residue, Nonfilterable (TSS)	mg/L	36	5.5	8
Secchi-Transparency	inches	13	55	31
Turbidity	NTU	43.0	3.23	7.99

Dissolved nutrient (orthophosphate and nitrate) concentrations are below detectable levels in all of the samples (Table 34). However, the detection limit for orthophosphate is somewhat high, in that the level of concern in lake eutrophication management is about 0.01 mg/L.

EPA (2001) recently proposed draft nutrient criteria for lakes and reservoirs. The nutrient criteria are developed based on Ecoregions. The North Platte Basin in Nebraska is located in Ecoregion V, South Central Cultivated Great Plains, and Subecoregion 25, Western High Plains. Reference conditions at both the Ecoregion and Subecoregion level are reported in EPA (2001). The reference for this region and subregion are shown in Table 35. The reference conditions are based on the 25th percentile of the data for each of the parameters in the EPA database. The 25th percentile was chosen based on studies that indicated that the 25th percentile of all data was equivalent to the 75th percentile of pristine sites. This has been called into question, and the nutrient criteria remain in draft at present.

Table 35. Value of nutrient criteria related data for Aggregate Nutrient Ecoregion V and subregion 25

Nutrient Parameters	Reference Conditions	
	Region	Subregion
Chlorophyll a (µg/L) (spectrophotometric method)	2.3	2.4
Secchi (m)	1.3	1.5
Total nitrogen (mg/L) (calculated)	0.56	2.42
Total phosphorus (µg/L)	33	24

Chlorophyll concentrations in Lake Minatare in 2004 range from 3.4 to 6.6 mg/m^3 (= µg/L). Even the lowest of these concentrations exceeds the reference conditions in Table 35. For a more valid comparison, some selected statistic of the data from Lake Minatare should be compared to the reference condition, but since the minimum is greater than the reference condition, it would not make any difference if this were the case. It would require recent data from other years to make such a comparison. The EPA database includes data from the period 1990 through 2000. The Lake Minatare data in Table 35 are more recent than this.

There is also considerable variation within the EPA database. Much of this variability is due to seasonal effects. A comparison between the EPA 25th percentile chlorophyll data for subregion 25 and the Lake Minatare data is shown on Figure 122. The fall sample shown for Lake Minatare is actually the late summer (August 18) sample. It is included as a fall sample to show the trend in EPA data from summer to fall. The EPA data show a peak in the chlorophyll concentration in the summer, while the Lake Minatare data show the minimum in 2004 in the summer. Alternatively, the maximum Lake Minatare chlorophyll in 2004 occurred in the late summer. The comparable Lake Minatare summer chlorophyll would be somewhere between the two concentrations shown on Figure 121, *e.g.* 5 µg/L or approximately the same as the EPA value of 5.3 µg/L. However, the Lake Minatare value of 5 µg/L would be equivalent to a median, rather than the 25th percentile. As noted above, EPA intended

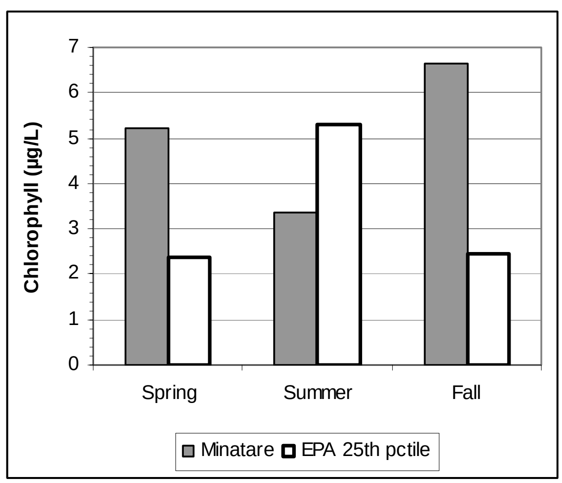


Figure 122. Comparison between Lake Minatare chlorophyll and EPA seasonal 25th percentile for subregion 25

using the 25th percentile as representative of an uncontaminated or natural representative site for the region and thus appropriate for a nutrient criterion.

Figure 123 is prepared in the same manner as Figure 122, but the EPA median chlorophyll concentrations are shown. Note that the seasonal trend in the EPA medians is different from the 25th percentiles shown earlier. There is now a decreasing trend from spring through fall. This still differs from the Lake Minatare trend. However, the Lake Minatare chlorophyll concentrations are well below the medians in the EPA seasonal database. If the EPA draft criteria are adopted as currently proposed, Lake Minatare would exceed those criteria and any water quality standards on which they are the basis. If there is some other way of interpreting the relationship between the monitoring data and such nutrient standards, then lakes like Lake Minatare may not be prime candidates for the list of impaired waters (see later section of this appendix on impaired waters).

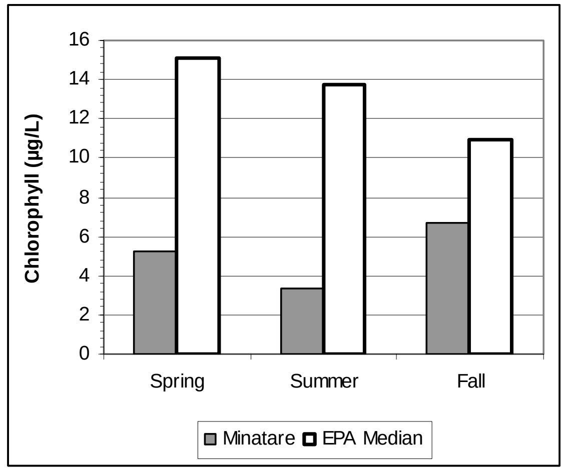


Figure 123. Comparison between Lake Minatare chlorophyll and EPA seasonal median for subregion 25

The Secchi depth criterion is related to turbidity, TSS, and chlorophyll. Chlorophyll is a surrogate measurement for phytoplankton (suspended or floating algae). The assumption in EPA (2001) is that the majority of suspended solids in lakes and reservoirs consists of phytoplankton, which are thus the major cause of turbidity. The Secchi depth is also a measure of water clarity, as is turbidity. Although most criteria are based on the 25th percentile, the Secchi depth criterion is based on the 75th percentile. The rationale is that while lower values are better for most standards, the Secchi depth increases as conditions improve. Consequently, the 75th percentile would represent better conditions than the 25th percentile. EPA chose the same rationale for a DO criterion, but no DO criterion is proposed with the nutrient criteria. A DO criterion has been in existence since EPA began to develop water quality criteria.

Figure 124 shows a comparison between the Lake Minatare Secchi depths and the EPA 75th percentiles from their database. The Lake Minatare Secchi depth data are in inches in Table 34. These are converted to meters on Figure 124. The overall Secchi depth criterion for Ecoregion V is 1.3 meters, while the subregional 75th percentile is 1.5 meters (Table 35), which is also the summer value on Figure 123. Each of the Lake Minatare Secchi depth readings is below the EPA criterion, as well as below the respective seasonal subregional 75th percentiles.

The pattern of the EPA 75th percentile seasonal Secchi depths show an increase from spring through fall. The Lake Minatare data show a peak in the early summer (July). The seasonal distribution of Secchi depth readings is not entirely consistent with the phytoplankton being the primary source of turbidity. The minimum Secchi depth occurs in the spring, when the phytoplankton (as represented by chlorophyll) are intermediate, but the turbidity and TSS are at their highest (Table 34). Alternatively, the Secchi depth may be controlled by phytoplankton density in the late summer, at which time the chlorophyll is at its highest, while turbidity and TSS are relatively low.

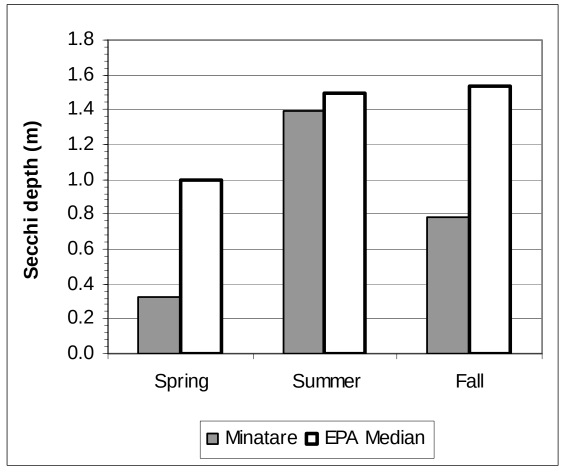


Figure 124. Comparison between Lake Minatare Secchi depth and EPA seasonal 75th percentile for subregion 25

The EPA (2001) proposes a criterion for total nitrogen. There are no total nitrogen data from Lake Minatare in 2004. Although total nitrogen can be calculated from the sum of nitrate plus nitrite (NO_3+NO_2) and total Kjeldahl (ammonium + organic nitrogen) and both source appear in Table 34, total nitrogen cannot be calculated because too many of the values are below their reporting limit. Nevertheless, something of a comparison with the EPA (2001) Ecoregional and subregional criteria can be made. The censored data from Table 34 can be used to estimate maximum or minimum concentrations of total nitrogen for comparison with the criteria. For example, the maximum total nitrogen concentration in the summer in Lake Minatare is <0.55 mg/L, while the Ecoregional criterion is 0.56 mg/L. There is no way that the Ecoregional criterion could be exceeded. Alternatively, in both the May and August samples, the NO_3+NO_2 is below the reporting limit, but the total Kjeldahl concentration is not. The total Kjeldahl concentration alone is greater than the criterion is each of those months. Consequently, the criterion must be exceeded in each of the months. Alternatively, the EPA (2001) subregional total nitrogen 25th percentile in Table 35 is 2.4 mg/L. None of the Lake Minatare total nitrogen concentrations would exceed that. The maximum that the total nitrogen concentration could be in each of the 3 sets of samples would be < 0.9, < 0.55, and < 0.73 mg/L, all of which must also be < 2.4 mg/L.

The last of the proposed EPA (2001) nutrient criteria is for total phosphorus (Table 35). If the Lake Minatare total phosphorus concentrations are converted to µg/L, (110, 50, and 60 µg/L), all of them exceed the Ecoregional and subregional reference concentrations of 33 and 24 µg/L respectively.

As was noted in the above section of this appendix on Pathfinder Reservoir, many factors in reservoir water quality can be related to total phosphorus loadings, but the direct effects of loadings are also affected by numerous factors as well. One of the major factors that affect how a lake or reservoir responds to excessive loadings is its flushing rate. A high flushing rate removes nutrients from the system and negates their availability for internal loading from the sediments. As was shown by the Pathfinder Reservoir hypolimnetic DO study, high total phosphorus loadings contribute to hypolimnetic DO depletion, one of the major consequences of eutrophication. Lake Minatare does not have significant hypolimnetic DO depletion based on the 2004 data. DO depletion can be visualized by the difference between the surface and bottom DO concentration in its profiles on Figure 121. These differences amount to about 1 mg/L in May and August and 2 mg/L in July. The profiles on Figure 121 also indicate that the reservoir experiences significant late summer drawdown. Although this is detrimental to the fishery

(Grier, 2005), it is also a factor in removing nutrients from the reservoir, as well as a factor promoting vertical mixing and aeration of the water column.

Effects of Alternatives

The only alternative that could potentially affect the Inland Lakes is the Full Water Leasing Alternative. The lakes are part of the North Platte Project, supplied from the Interstate Canal. The alternative as formulated for the FEIS envisions leasing a significant amount of the water from the North Platte Project supply. This should reduce the amount of water delivered to the Inland Lakes. As is noted in the previous paragraph, one factor determining how the productivity of a lake or reservoir responds to a given nutrient load is the flushing rate. A decrease in the water supply to the Inland Lakes would decrease the flushing rate and increase the productivity in the lakes.
Data from Lake Minatare are presented above. The indications are that Lake Minatare is already somewhat eutrophic (overly productive). This conclusion is based on the presence of elevated phosphorus and chlorophyll concentrations in the reservoir.

Another indicator of eutrophication is DO depletion. As was noted above, there is some degree of DO depletion in Lake Minatare. DO depletion is measured by comparing the measured concentration to a calculated saturation concentration. The calculated percent DO saturation for each of the 2004 Lake Minatare profiles is shown on Figure 125. As is indicated on the figure, all of the measurements are below 100 percent saturation. This result would indicate that total respiration in the water column exceeds primary production. The greatest depletion is shown by the August profile, in which all measurements are less than 75 percent saturation.

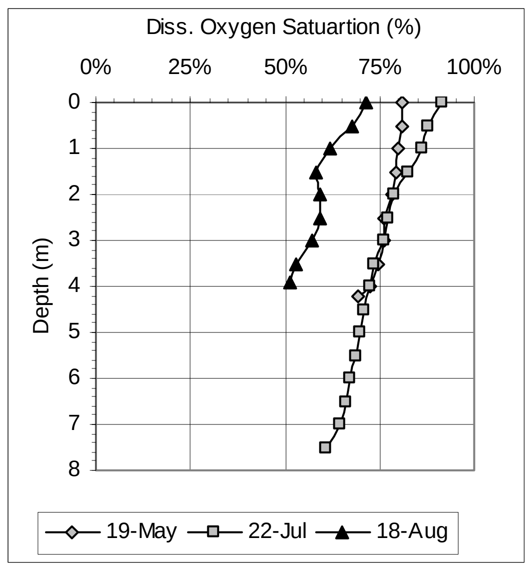


Figure 125. Percent dissolved oxygen in Lake Minatare on 3 dates in 2004

The current EPA criteria for DO for warmwater aquatic life include a 30-day mean, 7-day mean minimum, and a 1-day minimum (EPA, 2002; 1986). The 30-day mean is 5 mg/L, the 7-day mean minimum is 4 mg/L, and the 1-day minimum is 3 mg/L. The 7-day and 1-day minima represent the low points in the diurnal DO cycle (see earlier section on the North Platte).

The Full Water Leasing Alternative explores the effects of large-scale leasing of water in each state. To obtain such a large amount of water through leasing, it would be necessary for significant changes in state water laws and water administration arrangements. Under this alternative, and with these assumptions, it would be possible that a significant amount of water might be leased from irrigation districts in the Scottsbluff area. It is likely that part of the leasing would be from the Intestate Canal, but some of the leases would also be in districts served by the Fort Laramie, Tri-State, and Northport canals, and possibly others. Only leases in irrigation districts served by the Interstate Canal would affect the inland lakes.

State laws in both Wyoming and Nebraska provide that only the consumptive use portion of a water supply can be leased and transferred to another use, in order to protect local return flows on which other water users may depend. These requirements minimize the impact of leasing on local stream flows and ditch flows. However, if water is leased from lands near the coldwater streams in this area, there may be some reduction in irrigation runoff to the streams due to the practical limitations in achieving the requirement that stream flows not be affected when water is leased. The amount of reduction cannot be estimated absent a specific plan for water leasing, which would be based on voluntary participation from water users. The likely effect would be a reduction in some flow volume. The North Platte River EIS Model (NPREIS) made some projections based on various assumptions about where the leasing might occur. Changes in deliveries based on the model output are shown on Figure 126.

Deliveries to the Inland Lakes are made in April and October. The lakes are filled in April, and partially filled in October. The total capacity of the Inland Lakes is about 72 KAF. The most dramatic change is projected to occur under the water supply conditions experienced in the early 1950s (Figure 126). The reduced delivery under those conditions could amount to nearly ½ of the total capacity of the lakes. Because the Inland Lakes were not operated separately in the NPREIS, changes in individual lakes cannot be shown. However, the water passes through the system and the effect would likely to be greatest in the largest lake. As is shown on Figure 126, there is no change in the amount of water delivered to the lakes in most years. Based on the model output, there would be no change in deliveries to the Inland Lakes in 32 of the 48 years in the model.

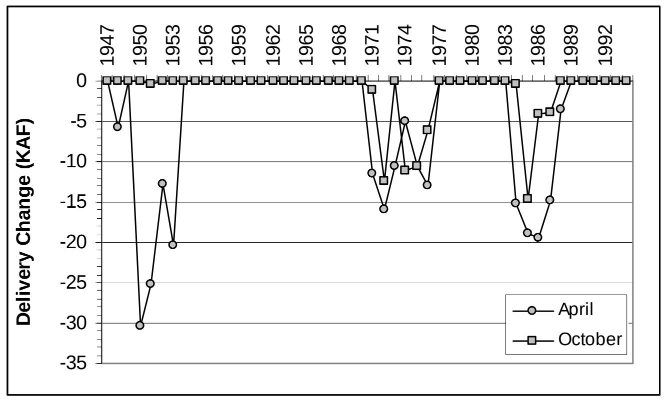


Figure 126. Change in deliveries to the Inland Lakes with the Full Water Leasing Alternative

Although the reduction in deliveries would reduce the flushing rate in the Inland Lakes, it would also reduce the nutrient load to the lakes as well. The water that is delivered to the lakes is diverted at the Whalen Dam near Torrington, Wyoming. There are few water quality data available at the diversion, but the NDEQ collected total phosphorus samples at the Wyoming-Nebraska State Line in 2001 and 2002. Based on the assumption that the State Line data are reasonably representative of conditions at the diversion, the State Line data are plotted on Figure 127. The total phosphorus concentrations in the river were extremely high in both years. If the analysis is related to the types of conditions evaluated in Seminoe and Pathfinder reservoirs above, the reduction in the total phosphorus loads to the Inland Lakes during years when deliveries are curtailed may be more than enough to offset any reduction in flushing. To put things in perspective, the desirable concentration in the lake would represented by the minimum shown on the *y*-axis or 0.01 mg/L. The main loading would occur in April, when the concentration in the river is over 10 times the desirable concentration (Figure 127). At these concentrations of total phosphorus, the reduction in loading would still leave the load in the eutrophic range. However, the reduction in the phosphorus load would be more than 75 percent when the changes in inflows are the greatest. Consequently, the net effect on reservoir productivity would probably be minimal.

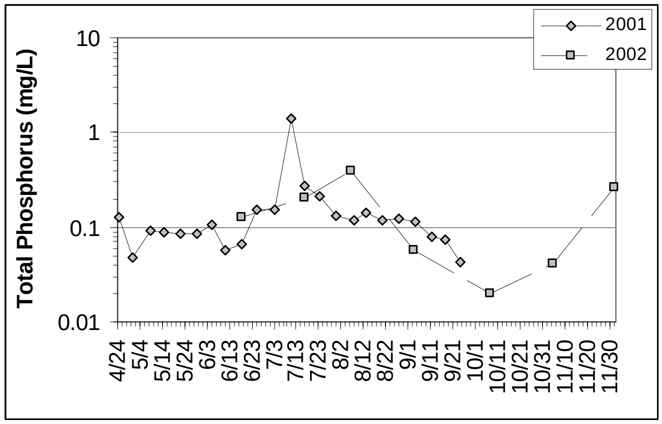


Figure 127. Total phosphorus concentrations in the North Platte River at the Wyoming-Nebraska State line

South Platte River

The Program elements in the South Platte River basin include the Tamarack element and water leasing in the basin between Fort Morgan and the State Line. The Tamarack element has 3 levels of development with each higher level including the lower ones. The Governance Committee alternative and the Program Water Emphasis alternative include the Tamarack phase III element, which would be an expansion of the phase I development and would include all phases. The Wet Meadow Emphasis Alternative would only include the phase I Tamarack development. There is a certain amount of water leasing in each of the alternatives. This section will evaluate the effects of the Tamarack operation. At this point, the water leases involve water stored in existing reservoirs. Under existing conditions, this water is primarily released to the river and leasing would only change the point of delivery. Effects on water quality in the South Platte River are not anticipated.

The Tamarack element involves the diversion of water from the South Platte River when unappropriated water is available. The water is applied to a series of recharge pits adjacent to the river. The ground water subsequently contributes to the base flow of the river. The effects will relate to the addition of ground water of possibly different quality back to the river.

Affected Environment

For purposes of this environmental assessment, the affected environment will include the South Platte River in the vicinity of Julesburg, Colorado. The South Platte River at the Julesburg gage is relatively saline. The EC has averaged about 1,820 µmho/cm at the Julesburg gage over the period of record used in the hydrologic models for the Recovery Program (Figure 128). Over the same period the TDS has averaged 1,270 mg/L at Julesburg. The diversion of water from the South Platte River immediately upstream from Julesburg and its subsequent return has the potential to change the EC of the river at Julesburg by increasing the evaporation and consequently further concentrating its TDS. This section of the report will evaluate the possible effect of the diversion of South Platte River water and its subsequent return on its TDS as measured by EC.

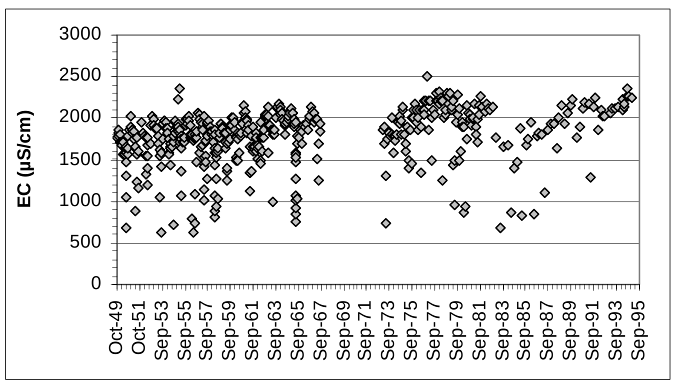


Figure 128. Specific conductance data for the South Platte River at Julesburg for the period of record 1951-95

There has been a significant trend of increasing EC over the period of record at the Julesburg gage. The EC data are plotted on Figure 128. A trend analysis showed a probability associated with the trend line of less than one in a 100 million of the trend being due to random chance. The trend is dominated by the EC data measured since the 1970's. The early data show much more scatter and an apparent greater degree of randomness in their distribution.

An extensive study of the ground water of the Tamarack site was conducted by Burns (1983; 1985). Burns (1983) established an observation network of 57 surface and ground water monitoring sites on and adjacent to the Tamarack Ranch State Wildlife Area (Table 36; Figure 129). The purpose of the data collection was to aid in an evaluation of water development projects on the wildlife area, particularly in relation to augmenting the development of what are known locally as "warm water sloughs." The purpose of this analysis is to evaluate the possible effects on water quality of such development. Therefore only sites with specific conductance data are of interest. The elimination of sites without EC data reduces the data set by nearly half (Table 36).

Table 36. Summary of Burns (1983) monitoring sites

Category	Count
Total sites[1]	57
Total wells	47
Sites with EC data	32
Wells with EC data	23

[1] Surface water sites include 5 in the river, 4 in sloughs, and 1 in a pond.

Table 37 shows the characteristics of each of Burns (1983) study sites. The vast majority of the sites were wells (Table 36); well locations are shown on Figure 129. Of the surface water sites the sloughs and the pond were not entered into the data base used to evaluate the Tamarack element. There were no location data for the sites to use in a definitive evaluation against local ground water. If the operation contemplated in the Tamarack Plan had been in operation, the water quality of the sloughs would have reflected that of the ground water returns to the river. Daily flow and monthly water quality data from the Julesburg gage on the South Platte River were retrieved to supplement the river data from Burns (1985) five river sites. Because flow measurements associated with the water quality data were few, the Julesburg daily flow data provided the only opportunity to evaluate the river flow against the water level data in the wells, despite the fact that the gage is over 200 feet lower in elevation and a considerable downstream from Burns study area.

Table 37.--Statistics of specific conductance by groups (Burns, 1985)
[in microSiemens per centimeter at 25 ° Celsius]

Group	Mean	Standard deviation	Minimum	Maximum
Wells in the sandhills	264	28	180	350
Wells in the valley meadow	712	383	260	1,800
Wells in the river bottom	1,160	404	370	2,150
Sloughs	1,040	402	360	1,600
River	1,540	237	700	2,000

Burns (1985) divided the well locations into three categories. The sandhills sites are generally located farthest from the river. However, those sites occupy the areas of greatest permeability and are most suitable for recharge. The valley meadow sites occupy low terraces adjacent to the alluvial sites of the river bottom. As can be seen from the EC data associated with the various locations, there is considerable variation in the ground water EC both among and within the various well locations. In general, the EC increases with decreasing distance to the river. The river EC is considerably higher than that of the ground water on the site.

Burns (1985) also constructed a ground water model of the Tamarack site. Although the model did not include a water quality component, Burns (1985) used the model results to estimate the changes in EC. Using the model outputs, a mass balance calculation of the EC of the Tamarack discharge was developed. A relationship between the EC of the discharge in relation to that of the recharge was developed.

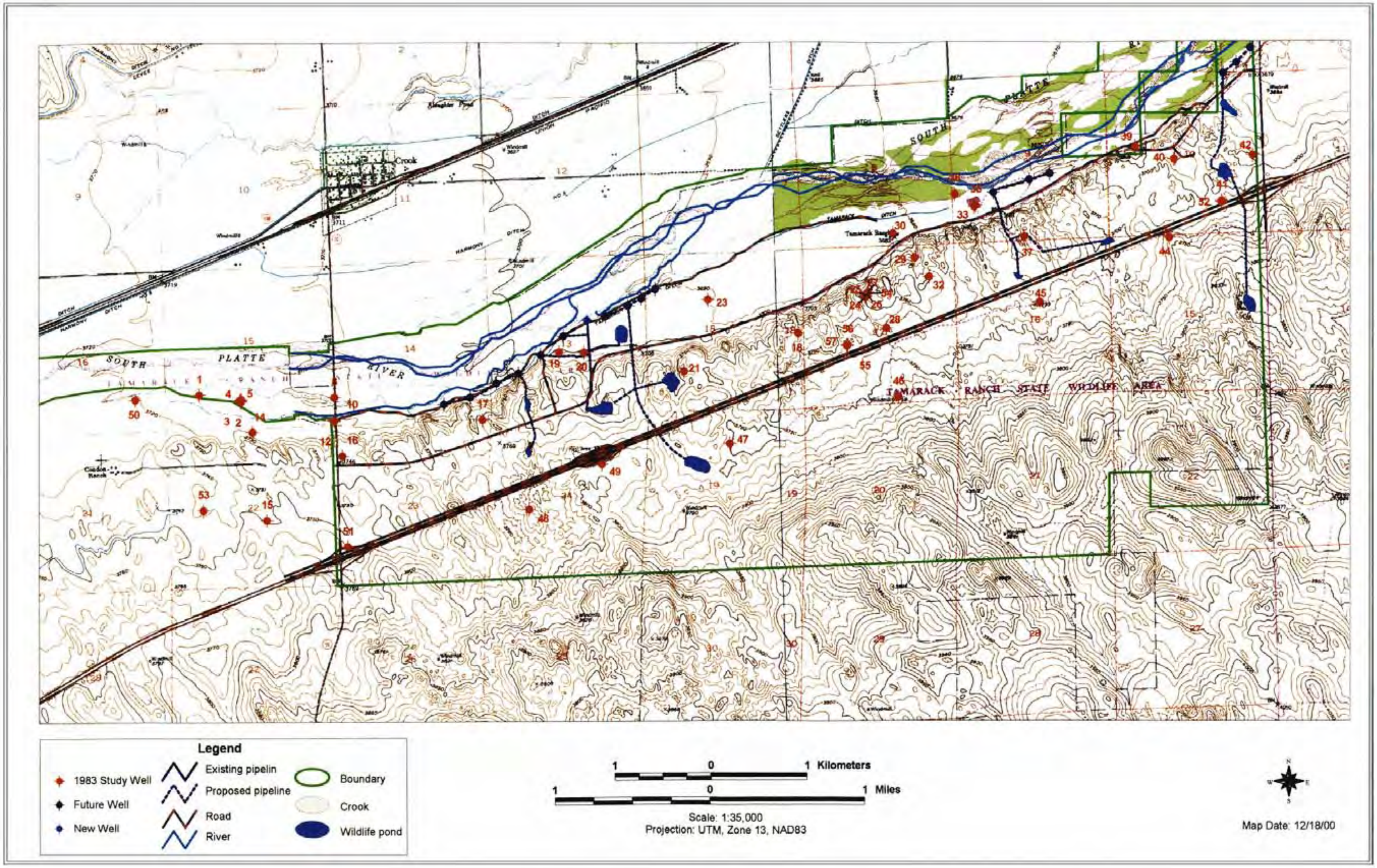


Figure 129. Location of Burns (1983) wells on the Tamarack Wildlife Area

Effects of Alternatives

Method of analysis

A number of different methods were attempted to develop an EC-flow relationship for the Julesburg gage data. As was the case of the North Platte River at Lisco, the relationship was generally poor. The best relationship was based on a subset of the data ($r^2 = 0.42$), using only EC data associated with instantaneous flow measurements. The relationship was greatly improved with the addition of a measure of "season". The observed and predicted EC data, along with the multiple regression equation and its R^2, are shown on Figure 130. The "season" used in the multiple regression is based on time of the year, and is not exactly the seasons of the year. The "seasons" were numbered from 1 through 3, with winter assigned a value of 1 and summer a value of 3; spring and fall, which are transitional seasons, were each assigned a value of 2. Although there is considerable scatter in the data at higher EC values, the regression is at least as good as some of the North Platte TDS-flow regressions.

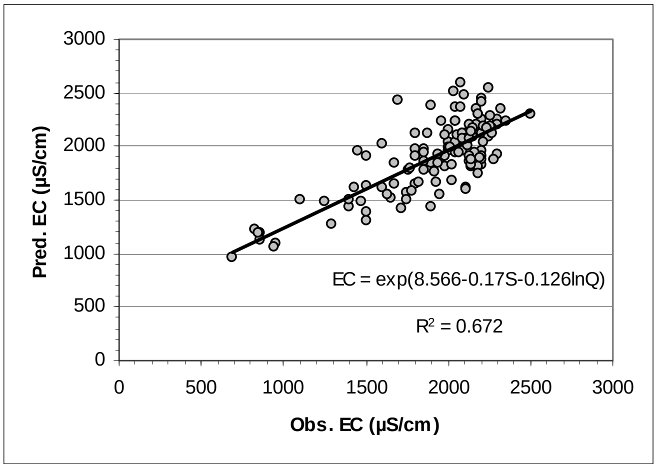


Figure 130. Multiple regression of EC on flow and season

The relationship on Figure 130 will be used to estimate the EC of the river at Julesburg for the Present Condition. To evaluate the effects of the alternatives, the recharge was taken from the Central Platte EIS Operations Model output, based on operations data from the Governance Committee's South Platte Operations Model. The EC was calculated based on the net recharge/discharge from the model. If the net diversion of a given alternative was negative, recharge will be assumed and the EC of the river associated with each respective alternative does not change from the Present Condition. If the net diversion of the alternative was positive, the EC of the Present Condition was adjusted by the flow and EC of the Tamarack discharge.

As was noted in Burns (1985), the effect on EC of the recharge will be due to evaporative concentration. The discharge will also be affected by mixing with the water in the aquifer. Burns (1985) developed estimates of the EC of the recharge and discharge based on a mass balance from his ground water model. The relationship is shown on Figure 131. The relationship shown on Figure 131 will be used to estimate the EC of the Tamarack water entering the South Platte River. In general, if the EC of the recharge is low, the EC will increase, and if the EC of the recharge is high, the EC will decrease. For the most part, the

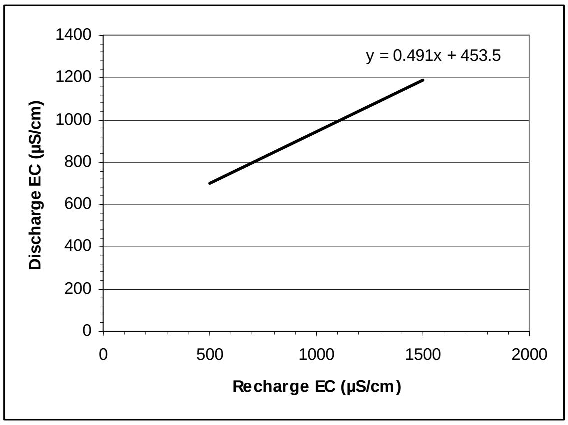


Figure 131. Burns (1985) recharge/discharge EC relationship

EC of the discharge from the Tamarack Project should increase, based on the usual EC of the river.

Present Condition

The Present Condition includes some development at the Tamarack site. The historic EC data shown on Figure 128 also included some level of development late in the data set. More recent EC measurements have been made by the NDEQ at the Julesburg gage (the NDEQ Stateline site). The monthly NDEQ data from 2002 and 2003 are shown on Figure 132. The EC ranged from about 1,200 to 2,500 µS/cm during that period. The lowest EC readings occurred during the winter, while the peak EC occurred during the summer. The flow at the gage during the summer is dominated by return flows from irrigation. During the winter, the river consists of base flow, which is apparently lower in EC than other flows in the river. The EC shown in Table 37 for the sloughs represents ground water gains from the Tamarack site. The EC of the sloughs is similar to that of the river during the winter, when base flow dominates.

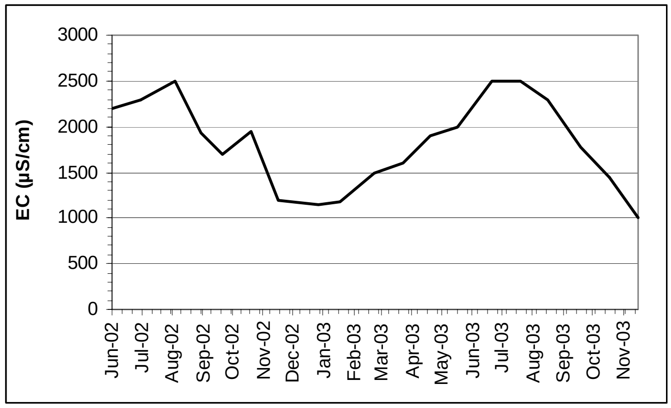


Figure 132. NDEQ specific conductance data for the South Platte River at Julesburg during 2002 and 2003

As was the case with the North Platte, data from 2002 and 2003 represent a drought condition. This raises the question as to whether the data represent the Present Condition or the effects of drought. Figure 133 shows the EC as developed from the multiple regression shown on Figure 130 and the Present Condition flow data. The range of EC shown on Figure 133 is generally the same as that on Figure 132, but the timing of the high and low monthly EC is different. The average EC for the Present Condition shows a very small range over the course of the year. The maximum and minimum monthly EC show lower EC during the summer, which was assigned the higher value for season. Despite the inverse relationship indicated for season above, the lower EC occurs with the higher season value. Based on this result, the EC from the multiple regression appears to be primarily dictated by flow, the other component of the multiple regression. From this perspective, the multiple regression should give a good basis for the impact analysis, which is greatly dependent on the flow model of the alternatives.

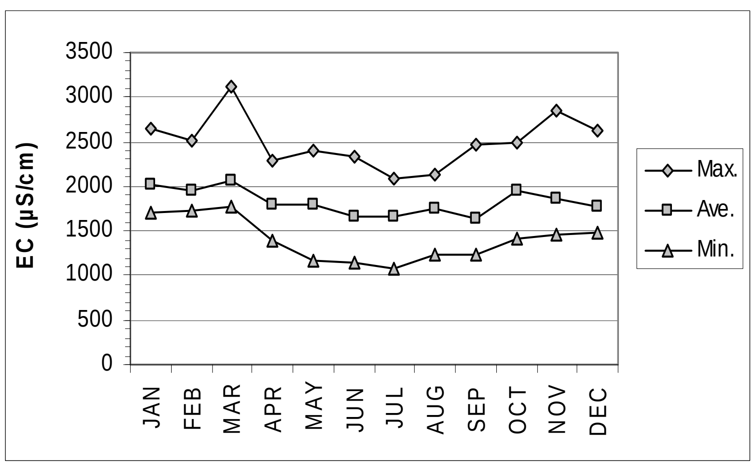


Figure 133. EC of the Present Condition at the Julesburg gage

EC of the Alternatives

Figure 134 shows the average EC of the Present Condition and each of the alternatives that include the Tamarack element. The Water Emphasis Alternative also includes water leasing, and the analysis includes a simulation of water leasing upstream from the Tamarack site. The Full Water Leasing Alternative does not include the Tamarack element and is not shown on Figure 134.

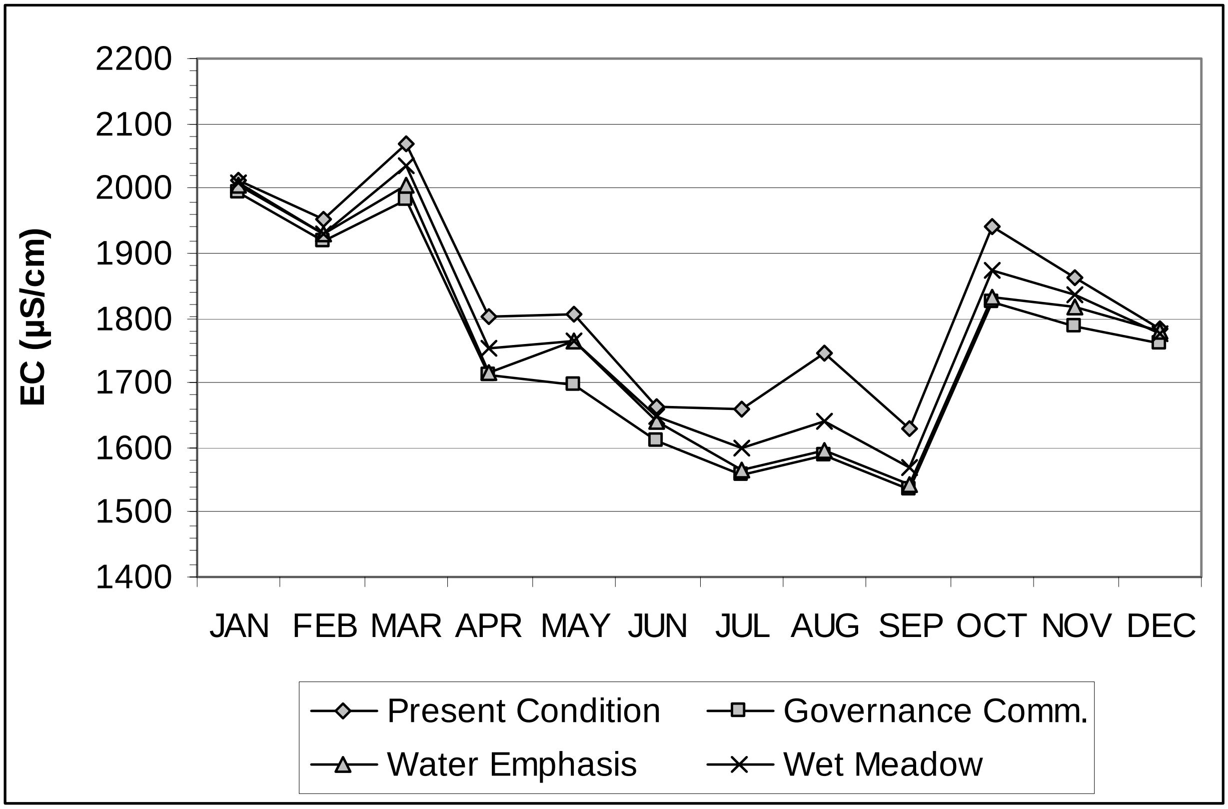


Figure 134. Average EC at Julesburg of the Present Condition and with the Tamarack operations under the various Platte River RIP alternatives

Figure 134 indicates that each of the three alternatives leads to a decrease in EC on the average relative to the Present Condition. The largest and most consistent decrease in EC would be under the Governance Committee Alternative, which includes Tamarack Phase III and no leasing. Consequently, the EC decrease is entirely due to the Tamarack operation. The smallest decrease in EC would be with implementation of the Wet Meadow Alternative, which includes only Tamarack I and no water leasing. The EC of the Water Emphasis is either intermediate between that of the other alternatives or equal to that of one or the other alternatives. The Water Emphasis Alternative includes both the Tamarack III and water leasing. The augmented flows associated with delivery of the leased water tends to offset the effect of the Tamarack III operations, which leads to the intermediate EC.

The EC remains relatively high at the Julesburg gage with any of the alternatives. Decreases are as small as 5 µS/cm in some alternatives and are generally less than 100 µS/cm. The maximum decrease during the summer is about 150 µS/cm, which seems large, but is less than 10 percent of the base EC at the Julesburg gage.

Water Quality of Lake McConaughy and Lake Ogallala

Affected environment

Lake McConaughy was classified as meso-eutrophic by the EPA (1976) in their National Eutrophication Survey in 1974-75. Phosphorus loadings during the study were more the twice the eutrophic level calculated in accordance with Vollenweider and Dillon (1974). Over 90 percent of the estimated phosphorus loading came from nonpoint sources in the North Platte Basin upstream from the reservoir (EPA, 1976).

The more critical water quality problem in Lake McConaughy relates to high temperatures and low dissolved oxygen, the latter of which is symptomatic of eutrophy. Lake McConaughy formerly supported a 2-story fishery. When the reservoir becomes thermally stratified in the summer, the surface layer supports a warmwater fishery consisting of walleye, white bass, catfish, smallmouth and striped bass (Van Velson, 1978). The coldwater fishery consisted of rainbow trout. The trout fishery in Lake McConaughy has been lost, but the cool hypolimnetic releases support a trout fishery in Lake Ogallala. Lake Ogallala is an afterbay downstream from Kingsley Dam. Lake Ogallala supports an important trout fishery, which is entirely dependent on cool releases from Kingsley Dam. If the releases from the dam are warmed by the program, then there would be a warming of Lake Ogallala as well. Such a warming would affect the trout fishery in Lake Ogallala. The coldwater fish habitat in Nebraska is defined as that with a temperature of 72°F or lower. The water with a suitable temperature is confined to the deeper waters of Lake McConaughy during the summer months.

Table 38 summarizes data on water quality of Lake Ogallala collected on several dates from 1989 through 1991. Unlike Lake McConaughy, whose deeper waters have historically undergone complete oxygen depletion during the summer, DO would not appear to be an issue in Lake Ogallala, based on the data in Table 38. However, that is not entirely true, Historically, the DO in the inflow to Lake Ogallala was adequate, even when the water was drawn from the anoxic hypolimnion of Lake McConaughy. Prior 1985, water passing through the control gates of Kingsley Dam entrained large quantities of air resulting in high DO concentrations in Lake Ogallala, but with the introduction of a hydropower facility, air entrainment has been greatly reduced (Stansbury et al., 2002a). The water of hypolimnetic origin in Lake McConaughy, although cool, has low DO and contains potentially toxic concentrations of ammonia and hydrogen sulfide, along with other oxygen demanding substances (*ibid.*). The effects of these problematic factors is further influenced by large beds of macrophytes downstream from the plant discharge, which are suspected of further depressing DO in the Keystone Basin of the lake near the outlet gates (Stansbury et al., 2002a; Dove *et al.*, 2002). As a result, the fishery in Lake Ogallala has been under stress and fish kills have been reported in August in the northeast corner of Lake Ogallala (Dove *et al.*, 2002). To date, an aeration system (Howell-Bunger valve) has been installed in the outlet of the hydropower facility (*ibid.*) and aerators that inhibit stratification have been installed in the North Basin (*ibid.*). Hoagland *et al.* (2000) presents a hypothetical rationale for the causes underlying the fish kills. In an attempt to mitigate the effects of the poor water quality on the fishery in Lake Ogallala, a series of studies have been undertaken to better define the problem and develop solutions (Admiraal *et al.*, 2003; Dove *et al.*, 2002; Stansbury et al., 2002a; 2002b).

Table 38. Summary of water quality in Lake Ogallala					
	Water Temperature (°C)	Secchi Disc Transparency (Inches)	Specific Conductance (µmhos/cm at 25°C)	Dissolved Oxygen (mg/L)	Dissolved Oxygen, Percent of Saturation (%)
Minimum	1.0	33.0	550	8.2	86.3
Median	17.5	45.5	746	10.5	110.0
Maximum	19.5	64.0	843	20.5	160.2
No. of Obs.	6	6	6	6	6

The Secchi depth, a measure of water clarity, has been relatively low, with a range of just under 3 feet (33 inches) to a little over 5 feet (64 inches — Table 38). Secchi depths of 40 meters (over 130 feet) have been recorded in extremely clear pristine waters, while Secchi depths of less than 3 feet are more characteristic of polluted (productive) water bodies (Cole, 1979).

Environmental Consequences

Method of analysis

Yahnke (1990) modeled the temperature and DO regime in Lake McConaughy. In the modeling study, in addition to the delivery of water from what would be the equivalent of variously sized environmental accounts, the effect of different water surface elevations in April on the ability of the Lake to meet the trout criteria in August were evaluated. There was a definable relationship. If the spring elevation of Lake McConaughy was at or above 3250 feet, the criteria were met in 4 out of the 4 years simulated. This dropped to 1 out of 4 years with a spring elevation of 3240 or 3230 feet and none out of 4 years at 3220 feet. Although the trout fishery in Lake McConaughy is no longer viable, these April elevations will be used as benchmarks in comparing alternatives. The temperature effects would still be important to the Lake Ogallala trout fishery. This is similar to the comparison to a flow of 1,200 ft^3/s in the Central Platte for the temperature impact analysis and is a relatively straightforward means of comparing alternatives.

The April elevations were extracted from the end-of-month elevation table in the hydrology model runs for the various alternatives. The number of years in the total record of the hydrology model (48 years) that the reservoir elevation in April was projected to be at or above 3250 feet were enumerated, along with the number of years that the April elevation was between 3240 and 3250 feet, and the number of years that the April elevation was below 3240 and below 3230. The comparison of these counts against those of the present condition should give an indication of the change brought about by each of the alternatives.

The characteristics of Lake McConaughy with respect to temperature and DO at the different April water surface elevations in each of the 4 years simulated in the model study are summarized in Table 39. The table includes the minimum thickness of the habitat layer as defined in Van Velson (1978), which consists of a temperature of 70°F and a minimum DO of 3 mg/L. The table also shows the depth of these layers, which in most cases coincide. This depth-of-layers category is only important if the depths of the layers do not coincide and therefore do

Table 39. Summary of minimum habitat or "worst case" combinations of temperature and DO in relation to different April water surface elevations

Elevatio	"Worst case" Condition	1974	1977	1978	1980
3250'	Minimum thickness (ft.) of $ 70°F & # 3 mg/L layer	< 1½	8	20	< 1½
	Depth of < 70° F Layer (ft.)	Absent	53-61	25-45	51-52
	Depth of > 3 mg/L layer (ft.)	56	53-61	25-45	51-52
	Duration of temperature > 70°F (days)	1	0	0	0
	Duration of DO < 3 mg/L (days)	0	0	0	0
	Maximum temperature (°F)	70.2	< 70	< 70	< 70
	Minimum DO (mg/L)	3	> 3	> 3	> 3
3240'	Minimum thickness (ft.) of $ 70°F & # 3 mg/L layer	0	0	15	0
	Depth of < 70° F Layer (ft.)	Absent	Absent	30-45	Absent
	Depth of > 3 mg/L layer (ft.)	Absent	Absent	30-45	Absent
	Duration of temperature > 70°F (days)	21	17	0	36
	Duration of DO < 3 mg/L (days)	5	9	0	15
	Maximum temperature (°F)	71.6	71.2	< 70	71.4
	Minimum DO (mg/L)	2.9	2.8	> 3	2.8
3230'	Minimum thickness (ft.) of $ 70°F & # 3 mg/L layer	0	0	5	0
	Depth of < 70° F Layer (ft.)	Absent	Absent	43-48	Absent
	Depth of > 3 mg/L layer (ft.)	Absent	Absent	43-48	Absent
	Duration of temperature > 70°F (days)	36	30	0	45
	Duration of DO < 3 mg/L (days)	20	16	0	21
	Maximum temperature (°F)	72.7	72.1	< 70	73.8
	Minimum DO (mg/L)	2.5	2.7	> 3	2.7

not define a layer of trout habitat, a condition that only occurred in 1974 (Table 39). The remaining 4 lines in Table 39 for each initial water surface elevation include the duration of any excessive temperatures or low DO and the maximum temperature when an excessive temperature occurs and a minimum DO if it falls below 3 mg/L.

The physical basis for the above effects relate to the mass of water, its relation to heat gain and loss, and the consequent distribution of DO in the reservoir. Heat gain in the reservoir is primarily at the water surface. The surface layer is warmed, while the bottom water remains cool. The density of water is temperature dependent, with warm water being less dense than cooler water, as long as the water is warmer than about 39°F. This distribution of heat in the reservoir causes a density stratification that isolates the cool denser bottom layer from the atmosphere. As long as the temperature induced density stratification is present, the deeper water cannot be aerated. Consequently the DO in the deeper layer cannot be replenished and is depleted as the summer stratification persists.

The ultimate late summer maximum temperature in the reservoir is related to the water surface elevation because a smaller mass of water exposed to the same conditions of heating will warm more than a larger mass of water. The April water surface elevation to some extent relates to the mass of water that will be present the following summer. Because of this factor, the April water surface elevation as shown in Table 39 also relates to the maximum temperature in the reservoir and in the transitional middle layer where the habitat exists. The smaller mass of water warms more and the warming is deeper in the lake than would be the case with a larger mass of water.

The smaller mass of water in the reservoir also results in a smaller mass of water in the isolated deep layer. This smaller volume of water contains a smaller mass of DO that can be more readily depleted than is the case with a larger mass of water in the reservoir. Therefore, the late summer deep water layer and at times the overlying transitional layer that constitutes the trout habitat has a lower DO concentration than would be the case in a larger pool, all other things being equal, as they were in the simulations that resulted in the data summarized in Table 2.

As is evident in Table 39, the temperature/DO simulations indicated that the trout habitat criterion that is the less likely to be met is temperature. The temperature criterion was exceeded by nearly 4 degrees Fahrenheit (73.8°F) at an April water surface elevation of 3230 feet in the 1980 simulation. The temperature greater than 70°F would have persisted for 45 days. The DO at the time would have been as low as 2.7 mg/L. However, the main purpose of Table 39 is to show the differences among years when the initial water surface elevations are the same. At 3250 feet, the habitat layer is present in all years except 1974. Table 39 does not indicate that the habitat layer has a thickness of 0 because there was a layer present with adequate DO that only exceeded the temperature criterion by 0.2°F for 1 day. There is enough uncertainty in the output (layer thickness of 0.5 meters or about 1½ feet and every 3 days) that the habitat could have been present but not output from the model as the output was defined by the output controls in the model code.

The only year in which the habitat layer remained present throughout the year at all 3 initial water surface elevations was 1978. When the April water surface elevation was set to 3220 feet, the temperature exceeded the 70°F limit for about 15 days; the maximum was 70.7°F. The minimum DO at the 3220 April water surface elevation was right at 3 mg/L. An April water surface elevation of 3210 was also evaluated using the 1974 data. The maximum temperature was 71.6°F, but the minimum DO was only 3.4 mg/L. The 70°F criterion was exceeded for 21 days with April water surface elevation set at 3210 feet.

To further illustrate the complexities of the model results, 1978 was slightly below average in terms of inflow. According to the operations model, the long-term average flow at the Lewellen gage is 1,420 ft^3/s. The mean annual flow in 1978 was 1,182 ft^3/s. However, the mean flow in 1977 was only slightly less at 1,137 ft^3/s, but showed some habitat loss, while 1978 did not. Alternatively 1980 was a near average year with a mean flow of 1258 ft^3/s, but the most persistent loss of trout habitat occurred in that year. The point is that not all of the effects can be directly related to flow. The interactions between the flow, reservoir pool, and climate are quite complex. However, for each of the years, the effect on the habitat becomes more severe in terms of both loss and duration as the April water surface elevation decreases. This latter effect is the basis for the Lake McConaughy comparative impact assessment among the alternatives.

Yahnke (1990) also evaluated the effects of decreasing water surface elevations on the peak release temperature from Kingsley Dam. The projected release temperature data are shown in Table 40. The same types of effects that were shown in Table 39 are shown in Table 40, which

Table 40. Comparison of Maximum Release Temperatures (°F) for Different Initial Water Surface Elevations (W.S.E.) in Each of the Four Simulated Years								
April W.S.E.	1974 Temperature		1977 Temperature		1978 Temperature		1980 Temperature	
	Peak	Date	Peak	Date	Peak	Date	Peak	Date
3260	66.6	9-Sep	67.1	13-Sep	66.0	11-Sep	66.0	8-Sep
3250	67.6	23-Aug	67.5	13-Sep	67.8	11-Sep	68.4	8-Sep
3240	70.9	23-Aug	71.2	26-Aug	68.2	28-Aug	70.2	25-Aug
3230	72.5	19-Aug	72.1	12-Aug	69.8	13-16	72.0	17-Aug

indicate that the increased warming extends to the deepest layers of Lake McConaughy. Actually the peak release temperatures are present when the reservoir undergoes a breakdown in the summer stratification, *i.e.* fall overturn or mixing. Consequently the duration that the temperatures would be present is relatively short because the reservoir should begin cooling following the onset of mixing.

Decreasing spring water surface elevations have a similar effect on release temperatures to those on trout habitat in Lake McConaughy. The lower water surface elevation not only relates to a higher peak release temperature, but to an earlier occurrence of that peak. Since the peak temperatures coincide with the breakdown of stratification, the earlier peak actually shows that stratification is shortened with reduced water surface elevations.

The temperature remains below 70°F in the Lake McConaughy release in all of the years when the spring water surface elevation is at or above 3250 feet (Table 40). At a spring water surface elevation of 3240 feet, the release temperature exceeds 70°F in 3 of the 4 simulated years, although the 1980 simulation only exceeds 70°F by 0.2°. At the 3230 foot spring water surface elevation, 70°F is once again exceeded in 3 of the 4 simulated years, but all of the peak temperatures are 2° or more over the temperature criterion for trout.

Another method of evaluating the release temperatures in relation to the Lake McConaughy is presented in the Lake McConaughy Fisheries Appendix. The analysis is based on a regression relationship between the Lake McConaughy water surface elevation and its release temperature. Given the limitations noted above, that method may give more reasonable results.

Lake Ogallala is a small reservoir. At full pool the reservoir capacity is 2,500 acre-feet. Active storage in Lake Ogallala is usually between elevations 3123.5 and 3126, the full pool elevation (Jerry Kirkman, Central Nebraska Public Power and Irrigation District, personal communication to Curt Brown, Platte River Endangered Species Recovery Program EIS Manager, August 15, 2002). The storage between the above elevations is 1,600 acre-feet and represents the normal regulatory capability of Lake Ogallala.

The residence time (the amount of time for inflows to completely replace storage) in Lake Ogallala is relatively short. Operation of Kingsley Dam would change with each of the alternatives. The different operations would change the releases from Kingsley Dam and change the residence time in Lake Ogallala in the process. Although the releases of warm water from Lake McConaughy may only last a day or so, this may be enough time to completely replace the storage in Lake Ogallala, causing it to warm to the release temperature. In cases where the release temperature is above that suitable for trout, adverse effects on the fish are possible in Lake Ogallala.

Because of the short residence times in Lake Ogallala, the evaluation of residence times is based on a conversion of the simulated monthly Kingsley Dam releases to an average daily flow for each month. There was no simulation of Lake Ogallala pool levels or volumes. To calculate the residence times, the July and August Kingsley Dam releases in acre-feet per day were mixed with Lake Ogallala volumes set at a constant level halfway between the minimum and maximum pool volumes in acre-feet. July and August were selected because of the uncertainty in the timing of mixing. In the Lake McConaughy temperature model, mixing occurred in August under all circumstances, but monitoring data indicate that mixing may occur in July under some circumstance. The Lake Ogallala residence times were calculated in days, which were also converted to hourly values by multiplying by 24.

Present conditions

The Present Condition has existed in Lake McConaughy since 1997, as a result of the operating conditions imposed in the FERC license. The NDEQ measured temperature and DO profiles in Lake McConaughy during 2004. Profile plots of the data appear on Figure 135. At the time that the profiles were measured, the water surface elevation was below 3220 feet (msl). Although the above description of the method to be used to evaluate the effects of the alternatives indicated that there was a high probability of bottom temperatures that may be unsuitable for trout, that was not the case in 2004. The maximum bottom temperature in the three profiles was 17.5°C (63.5°F). On the assumption that the release temperature would be drawn from a range of depths, a representative temperature for the release to Lake Ogallala would be 18.8°C (65.8°F). In either case, the temperature would be suitable for coldwater fish.

During May 2004, there was no significant thermal stratification in Lake McConaughy (Figure 135). The total depth of the May profile was 31 meters (102 feet). There was some degree of DO depletion below a depth of 10 meters (33 feet). In July 2004, there was relatively strong, but shallow thermal stratification. Coincidentally in July, DO depletion was extensive and tracked the thermal stratification. However, the minimum DO of 3.2 mg/L was observed in July 2004. By August 2004, stratification had deepened to a depth below 10 meters. Although there was a decline in DO that coincided with the thermal stratification, the bottom DO had recovered to near 5 mg/L (Figure 135).

The 2004 profiles on Figure 135 are considerably different from those used in the temperature-DO model described above. The model was based on data collected between 1973 and 1980. In that data set, the bottom DO went to or near 0 mg/L every summer, usually during August. Conditions conditions in 2004 are apparently considerably different from those when the data for the temperature model were collected in the late 1970s and early 1980s. Lake McConaughy was relatively shallow throughout 2004. The CNPPID website has a continuous record of Lake McConaughy water surface elevations since its initial filling. A copy of the plot of those water surface elevations is shown on Figure 136. As can be seen on Figure 136, the water surface

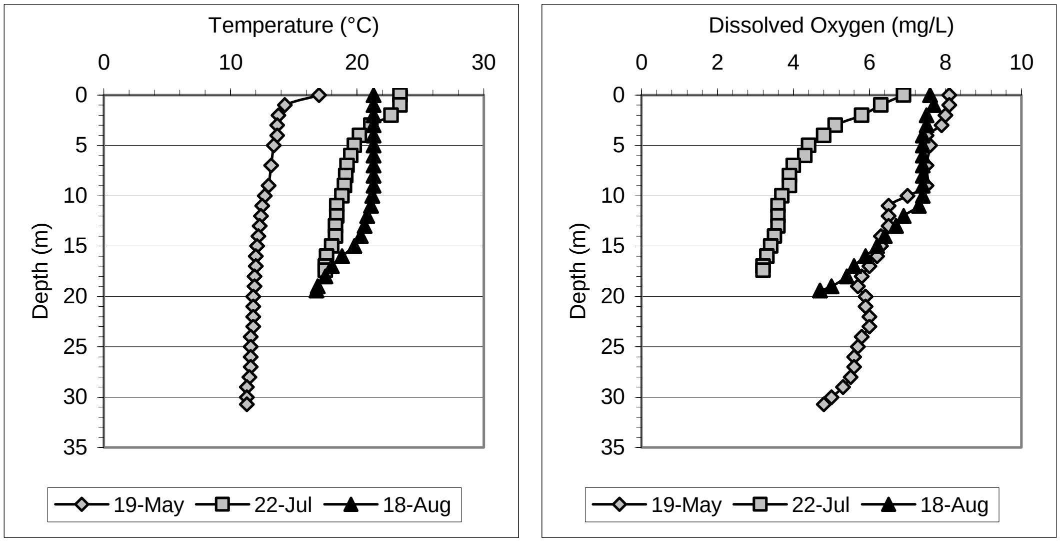


Figure 135. Lake McConaughy temperature and DO profiles in 2004 – Source: NDEQ

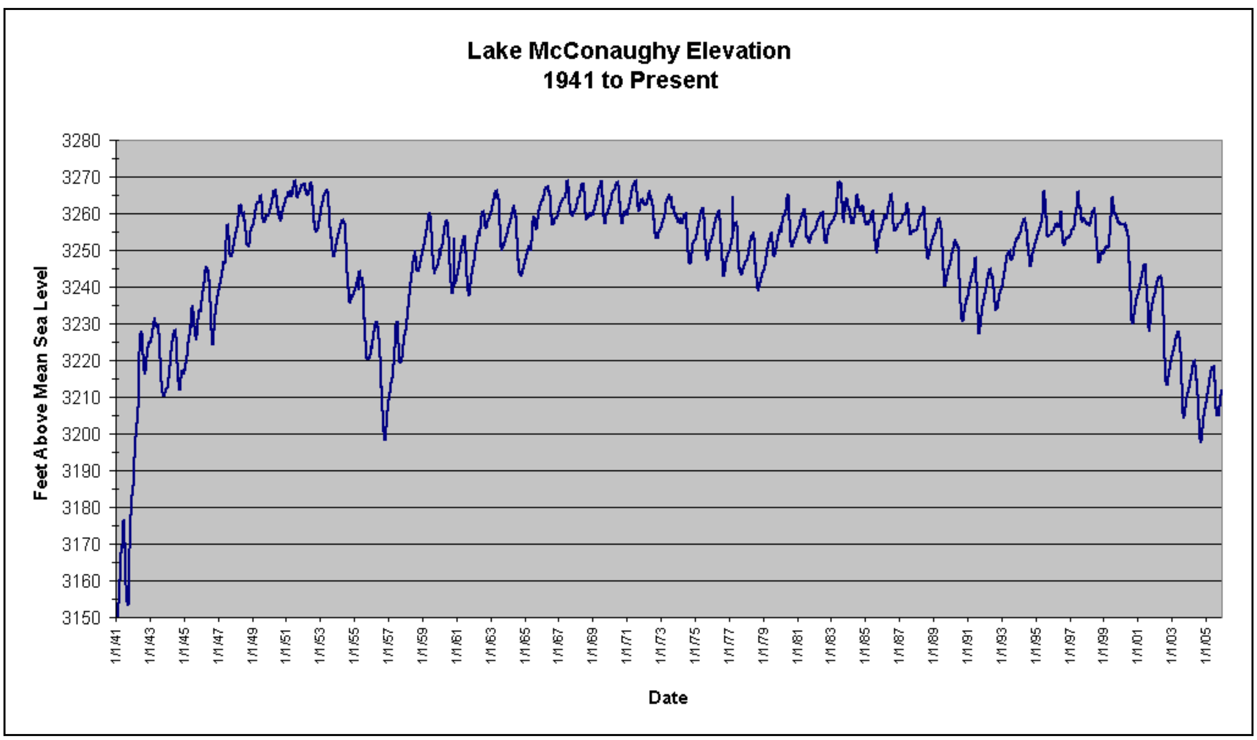


Figure 136. Lake McConaughy water surface elevation from 1941 through the summer of 2005 (Source: CNPPID)

water surface elevation has been declining since 2000. The water surface elevation of Lake McConaughy (and its content) reached a level about the same as its previous low point since its initial fill. That previous low pool occurred during the drought of the 1950s (Figure 136).

In the above discussion of the temperature/DO model, the relationship between surface area and insolation was indicated as the source of reservoir warming in the summer. Not mentioned was another source of heat to the reservoir, the inflowing water. In the model, the volume and surface area were decreased in the model exercise. However, the inflows and outflows were not changed. As a consequence, the heat in the inflow became increasingly important in the reservoir heat budget. In drought conditions, both the reservoir volume and the inflow would decrease and not necessarily in proportion.

Figure 137 shows a plot of the flows at Lewellen from 1965 through 2004. As has been noted before, the Lewellen gage represents the inflow to Lake McConaughy. The flows on Figure 137 are total annual flows in acre-feet. The numbers above each of the flows on the plot represent the rank of the year from low to high over the 40-year period of record used in the plot. The main point of the plot is to illustrate that the three years 2002 through 2004 rank as the three lowest inflow years to Lake McConaughy in the 40-year period. Under these circumstances, the amount of heat in the inflow would be reduced significantly, at least relative to the way the reservoir was modeled in Yahnke (1990). Because of the reduced delivery of inflow heat, the model results probably cannot be accurately applied to conditions such as those of 2002 through 2004.

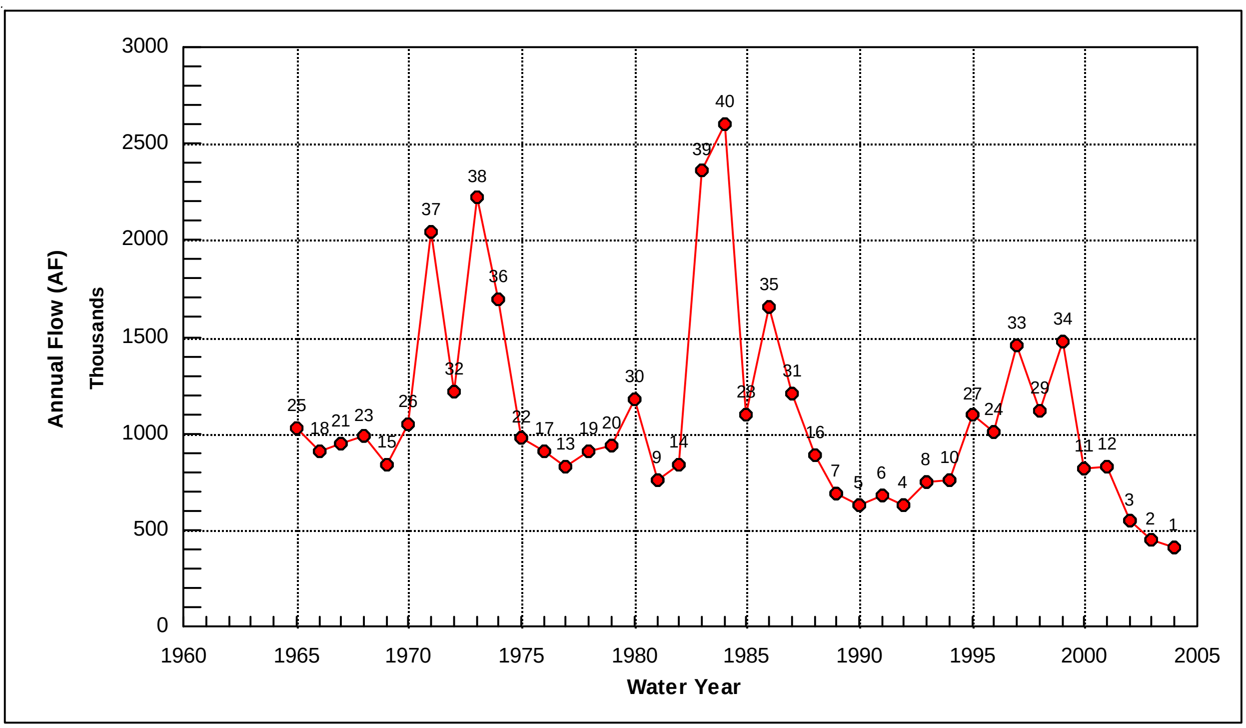


Figure 137. Total annual flow of the North Platte River at Lewellen from 1965 through 2004 (40 years)

As modeled in the Platte River EIS hydrologic model, under the Present Condition, the April Lake McConaughy water surface elevation is greater than 3250 feet in about 90 percent of the years. This result means that in about 10 percent of the years there is a chance that one or the other of the respective temperature or DO criteria would not be met, based on the years simulated. If the 4 simulated years are representative of a broad range of conditions, which the preceding admittedly brings into question, then the result could be translated to a probability, but the model results are such that it should only be reasonable to conclude that the probability that the criteria would not be met is somewhere between 0 and 75 percent in those 5 years out of 48 that the April water surface elevation is less than 3250 feet. Because the release water is aerated, the DO criteria are not really particularly meaningful. However, the elevated bottom temperature would be relevant to the Lake Ogallala inflow.

It is recognized that under historic operations, Lake McConaughy has fallen below 3230 feet, in particular in the last several years; these are not in the period included in the FEIS simulated operation. The reservoir also fell below 3230 feet during the drought of the 1950's, which is in the period of the simulated operation; this difference reflects a limitation in the capability of the hydrologic model to simulate reservoir operation.

The monthly frequencies associated with the spring elevations for the present condition that are described above also apply to the release temperatures in Table 40. The temperatures of water released from Kingsley Dam is well within the trout habitat temperature criterion of 70°F around 90 percent of the years during August, based on the monthly data.

The July and August water residence times range from 4.5 to 95 hours under the Present Condition (Table 41). It should be recognized that Table 41 represents a discharge to Lake Ogallala at ½ the active pool, as represented by a constant 1,700 acre-feet of storage. In other words Table 41 does not necessarily represent a particularly realistic condition. A more realistic operation would have the discharge to Lake Ogallala in a drawn down condition, ~900 acre-feet at the low end of active storage, then filled. Releases to the river would then be made from Lake Ogallala at some specified lower rate to meet downstream needs, drawing the reservoir down. Such an operation could extend the residence time in Lake Ogallala and allow for additional in-reservoir warming.

Table 41. Lake Ogallala Present Condition residence times (hours)

Month	Minimum	Median	Maximum
July	2.8	6.6	25.4
August	6.5	8.5	20.7

In addition, because of the morphometry of Lake Ogallala, complete mixing does not occur. There is a current that flows through the lake, while the area of the lake away from the Kingsley Dam outlet is not greatly mixed. The isolated area would warm more than the area most affected by the current. However, the isolated area is where aeration occurs further complicating flow patterns in the reservoir.

Under the circumstances assumed in Table 41, the rapid flow of cool water through Lake Ogallala would minimize warming of the water as it moves through the current-affected area of the afterbay. Alternatively, increased residence time would allow for further expression of the oxygen demanding substances and depress DO. This combination should maintain the trout population within Lake Ogallala during the warmest part of the summer during that 90 percent of the years that the spring water surface elevation is above 3250 feet. In the other 10 percent of the years, the inflow temperature would be within or near the trout temperature criterion about ½ of those years based on tables 39 and 40, and within about 1.2°F in the remaining years under Present Conditions.

Effect of Alternatives

Table 42 shows a comparison of the April water surface elevations in Lake McConaughy under the Present Condition and each of the alternatives. Table 42 indicates that the Governance Committee alternative would reduce the number of years in which the spring water surface elevation would be at or above 3250 feet from 90 percent to 73 percent of the years. In those years that the spring water surface elevation is reduced, peak release temperatures from Kingsley Dam would be greater than the trout criterion around ¼ of the time based on the variation among years when combining the 3240- and 3250-foot categories in Table 40. The effect would increase to ¾ of the time in the 10 percent of the years that the elevation would be at or below the 3240-foot spring water surface elevation.

Table 42. Comparison of Alternatives – percent of years that the April elevation is in each of the 3 categories characterized in Table 41

Elevation (feet m.s.l.)	Present Condition	Governance Committee	Water Emphasis	Full Water Leasing	Wet Meadow
> 3250	89.6%	72.9%	75.0%	91.7%	83.3%
< 3250 & > 3230	10.4%	27.1%	25.0%	8.3%	16.7%
< 3230	0%	0%	0%	0%	0%

The remaining alternatives, with the exception of the Full Water Leasing Alternative, show a decrease in the frequency with which the 3250-foot elevation is exceeded in April (Table 42). The Water Emphasis shows the largest decrease (15 percent) after the Governance Committee Alternative in the frequency of meeting the 3250-foot benchmark. Wet Meadow Emphasis Alternative shows decrease of about 6 percent in meeting the benchmark, but the 6 percent is the frequency that the water surface elevation would be below 3240 feet as well.. All of the alternatives, except the Full Water easing Alternative, but including the Present Condition, are projected to be below 3240 feet in the spring in at least some of the years. The Wet Meadow Restoration alternative, like the Governance Committee alternative (all scenarios, would be below 3230 feet during April in 3 of the years, while the remaining 2 alternatives would only be below the 3230-foot elevation in 1 year. Nevertheless, all of the alternatives increase the frequency that the April pool elevation will be below each of the benchmark elevations shown in Table 40.

Based on the Lake Ogallala residence time data in Table 43, the minimum, median, and maximum residence times would increase in July with all of the alternatives with the lone exception if the Full Water Leasing Alternative minimum, which would remain the same. However, the August residence times show variable results. The minimum residence times for all of the action alternatives would decrease. The median residence times of 3 of the 4 action alternatives would increase, while that of the Wet Meadow Emphasis Alternative would decrease. The maximum residence times of the Governance Committee and Wet Meadow alternatives would decrease, while those of the Water Emphasis and Full Water Leasing alternatives would increase.

Table 43. Comparison of Lake Ogallala residence time (hours) between the Present Condition and each of the action alternatives

Month	Alternative	Minimum	Median	Maximum
July	Present Condition	2.8	6.6	25.4
	Governance Committee	3.1	7.1	26.2
	Water Emphasis	3.1	7.7	31.5
	Full Water Leasing	2.8	8.0	31.8
	Wet Meadow Emphasis	3.1	6.7	30.1
August	Present Condition	6.5	8.5	20.7
	Governance Committee	5.6	8.7	19.4
	Water Emphasis	5.9	9.3	21.7
	Full Water Leasing	3.6	10.5	24.5
	Wet Meadow Emphasis	5.3	8.3	18.4

Although there are numerous increases in residence time in Lake Ogallala, most are no longer than an hour or so. An hour or 2 of additional residence time is probably not enough to make a great difference in trout survival, but it may be enough to bring about an increase in stress.

The remaining 3 alternatives have a dichotomy of effects on residence time in Lake Ogallala relative to the present condition. The Program Water Emphasis alternative shows an increase in the minimum and the median and a decrease in the maximum residence time. The Wet Meadow Restoration alternative shows the smallest increase in the minimum of any of the alternatives, the largest decrease in the median, and no change in the maximum residence time. The Water Leasing alternative shows the largest increase in the minimum and maximum residence times, but little effect on the median (Table 43). Nevertheless, the increased residence times at the upper end of the distribution by the Water Leasing alternative could exacerbate the DO problem in Lake Ogallala.

To summarize, the effects on the coldwater habitat of Lake Ogallala are affected by the effects on Lake McConaughy. The reduction in the April water surface elevation that results from each of the alternatives leads to warmer releases to Lake Ogallala in August, when the Lake McConaughy releases are warmest. The frequency of the release of water that exceeds the temperature desirable for coldwater habitat will increase with each of the alternatives, except for the Full Water Leasing Alternative. The greatest reductions in the frequency of meeting the 3250- and 3240-foot April elevations would be due to the Governance Committee Alternative, while the Water Emphasis and the Wet Meadow alternatives would have a lesser adverse effect. The Full Water Leasing Alternative would have a favorable effect and increase the percent of the years that the April water surface elevation was greater than 3250 feet.

Platte River Mainstem

Affected environment

A summary of all of the USGS monitoring data for 6 sites in the mainstem of the Platte River, except for pre-1975 selenium analyses, is presented in tables 44 through 46. The complete period of record for each site is also shown in tables 44 through 46. The summaries include the first and third quartiles (25th and 75th percentiles), the median (the 50th percentile), the minimum, maximum, and the number of observations. These are key parameters in the cumulative frequency distribution of the water quality data. These were used to compare with the water quality standards that existed at the time to flag potential indicators for the impact assessment. Another requirement for an indicator is an adequate database.

The data in tables 44 through 46 are presented primarily to summarize relevant data and to point out some of the complications in evaluating historic data. Current water quality problems in the Middle Platte Basin as defined by the NDEQ appear in a later section of this appendix. The total period of record for each of the sites is shown in Table 44. In each case the total period extends back beyond 1975. The main reason for using the 1975 cutoff relates to the selenium data. The USGS (1981) has issued a memorandum indicating that high values of dissolved and total selenium, *i.e.* greater than 5 µg/L (the current standard) from prior to 1975 were probably incorrect, as were values of dissolved selenium greater than 1 µg/L. For this reason the period of record for all standard comparisons were begun in 1975 for ease of data manipulation.

Tables 44 through 46 also include several water quality constituents for which there are no standards. These are either potential contaminants that are commonly used in the area (simazine and cyanazine) or are related to other constituents (specific conductance and TDS). The major complicating factor in making the comparisons over time is the detection limit for constituents such as arsenic and lead. The detection limit has varied over time. The main problem that this creates in a statistical summary is that some actual measured data are lower than earlier (or sometimes later) detection limits. This results in an intermingling of measured and unmeasurable results. An even greater difficulty arises when the standard is below the detection limit; this is a particular problem with lead and mercury. To avoid complications with the detection limit, the data were checked for flags that indicate that the value was less than a detection limit. Any values that were flagged as being less than the detection limit were assumed to meet the standard without checking further. These types of values may or may not meet the standard, but there is no way to evaluate that for sure.

The comparison between the water quality standards and data collected since 1975 was based on data no more recent than the 1990s. Based on the percent of the time that standards were exceeded, the greatest water quality problems in the mainstem of the Platte River were due to fecal coliform bacteria, arsenic, lead, and atrazine.

The fecal coliforms exceeded the standard of 400 colonies per 100 mL between 20 and 50 percent of the time depending on the monitoring site. As can be seen from Table 44, fecal coliform counts in the hundreds of thousands have been observed at 3 of the 5 sites for which

Table 44. Summary of water quality data at 6 sites in the mainstem of the Platte River in the Central and Lower Platte basins based on all samples collected during the period of record shown						
Location	Period of Record		Stream Flow (ft^3/s)	Water Temp. (°C)	Conductivity at 25°C (µmho/cm)	TDS (mg/L)
Brady	From:	Minimum	4	11.0	401	270
	10/23/56	25th Percentile	82	16.0	656	455
	To:	Median	184	20.0	793	539
	07/27/94	75th Percentile	605	23.0	1840	1312
		Maximum	3,720	29.5	1900	1520
		No. of Obs.	22	9	21	18
Overton	From:	Minimum	61	-0.1	421	270
	02/01/50	25th Percentile	500	4.0	840	550
	To:	Median	1,105	13.0	902	592
	09/13/94	75th Percentile	2,165	21.0	966	646
		Maximum	23,700	32.5	1235	776
		No. of Obs.	436	347	210	96
Grand Island	From:	Minimum	25	0.0	462	435
	09/17/64	25th Percentile	773	2.0	818	551
	To:	Median	1,430	14.0	879	582
	09/26/94	75th Percentile	2,660	22.0	954	628
		Maximum	19,200	30.6	1220	834
		No. of Obs.	241	239	220	146
Duncan	From:	Minimum	3	0.0	236	171
	11/08/45	25th Percentile	574	5.0	780	509
	To:	Median	1,350	16.0	845	566
	08/02/94	75th Percentile	2,460	22.0	916	616
		Maximum	16,900	37.0	1140	815
		No. of Obs.	352	374	362	262
North Bend	From:	Minimum	120	0.0	192	133
	09/18/64	25th Percentile	2,950	1.4	398	247
	To:	Median	5,000	13.8	480	299
	09/07/89	75th Percentile	8,300	22.6	575	364
		Maximum	73,600	37.0	880	560
		No. of Obs.	233	232	192	101
Louisville	From:	Minimum	390	0.0	100	168
	02/26/71	25th Percentile	3,120	8.0	484	347
	To:	Median	5,600	18.5	623	409
	03/17/95	75th Percentile	9,500	24.0	745	474
		Maximum	110,000	32.0	2250	995
		No. of Obs.	505	516	356	168

there are data. The coliform standard is based on water contact recreation use and does not relate to any of the endangered species or any other form of wildlife.

Table 44. Summary of water quality data at 6 sites (continued)							
Location		pH SU	$NH_3+NH_4^{-1}$ N Diss. (mg/L)	$NH_3+NH_4^{-1}$ N TOTAL (mg/L)	Un-ionized NH_3-N (mg/L)	Un-ionized NH_3-NH_3 (mg/L)	Fecal Coliforms (#/100 mL)
Brady	Minimum	7.30	0.02	—	0.0008	0.001	—
	25th Percentile	7.60	0.03	—	0.0009	0.001	—
	Median	7.70	0.03	—	0.0010	0.001	—
	75th Percentile	8.10	0.03	—	0.0015	0.002	—
	Maximum	8.40	0.03	—	0.0020	0.003	—
	No. of Obs.	21	3	0	3	3	0
	No. < D.L.	---	0	0	0	0	—
Overton	Minimum	7.10	< 0.01	< 0.01	< 0.0001	< 0.0001	0
	25th Percentile	7.90	0.02	0.04	0.0009	0.001	61
	Median	8.12	0.04	0.08	0.0020	0.002	190
	75th Percentile	8.42	0.07	0.12	0.0040	0.004	700
	Maximum	8.98	0.39	0.42	0.0240	0.03	190,000
	No. of Obs.	218	34	138	149	149	128
	No. < D.L.	---	3	14	3	3	—
Grand Island	Minimum	7.20	0.01	< 0.01	< 0.0001	< 0.0001	1
	25th Percentile	8.10	0.02	0.03	0.0006	0.0008	59
	Median	8.20	0.02	0.06	0.0020	0.0020	150
	75th Percentile	8.40	0.03	0.10	0.0040	0.0050	430
	Maximum	8.95	0.54	0.36	0.0390	0.0470	12,000
	No. of Obs.	234	25	192	215	215	154
	No. < D.L.	---	0	25	0	0	—
Duncan	Minimum	6.80	< 0.01	< 0.01	< 0.0001	< 0.0001	2
	25th Percentile	7.80	0.02	0.03	0.0004	0.0005	30
	Median	8.10	0.04	0.05	0.0010	0.0020	93
	75th Percentile	8.35	0.07	0.08	0.0030	0.0030	340
	Maximum	8.93	0.32	0.33	0.0380	0.0460	5,900
	No. of Obs.	366	122	76	141	141	101
	No. < D.L.	---	8	5	8	8	—
North Bend	Minimum	6.90	0.01	< 0.01	< 0.0001	< 0.0001	10
	25th Percentile	7.70	0.01	0.04	0.0005	0.0006	100
	Median	8.00	0.03	0.08	0.0010	0.0020	260
	75th Percentile	8.30	0.13	0.16	0.0040	0.0045	1000
	Maximum	8.90	0.59	0.81	0.0630	0.0770	280,000
	No. of Obs.	199	23	180	183	183	149
	No. < D.L.	---	6	16	6	6	—
Louisville	Minimum	6.05	< 0.01	< 0.01	< 0.001	< 0.001	10
	25th Percentile	7.97	0.02	0.04	0.001	0.001	140
	Median	8.28	0.04	0.10	0.002	0.002	370
	75th Percentile	8.60	0.11	0.22	0.004	0.005	1600
	Maximum	10.00	0.58	1.10	0.043	0.052	290,000
	No. of Obs.	369	195	147	270	270	149
	No. < D.L.	—	14	11	14	14	—

Table 45. Summary of inorganic contaminants at 6 sites in the mainstem of the Platte River in the Central and Lower Platte basins based on the period of record shown in Table 44						
Location		Arsenic As, diss. (µg/L)	Cadmium Cd, diss. (µg/L)	Chromium Cr, total (µg/L)	Copper Cu, diss. (µg/L)	Lead Pb, diss. (µg/L)
Brady	Minimum	< 1	< 1	—	< 1	< 1
	25th Percentile	—	—	—	—	—
	Median	< 1	< 1	—	< 1	< 1
	75th Percentile	—	—	—	—	—
	Maximum	< 1	< 1	—	< 1	< 1
	No. of Obs.	1	1	0	1	1
	No. < D.L.	1	1	0	1	1
Overton	Minimum	< 1	< 1	< 10	< 1	< 1
	25th Percentile	4	< 1	< 10	2	1
	Median	5	< 1	< 10	3	4
	75th Percentile	6	1	10	8	5
	Maximum	30	2	30	20	13
	No. of Obs.	14	10	31	15	11
	No. < D.L.	1	7	20	2	1
Grand Island	Minimum	< 1	< 1	10	< 1	< 1
	25th Percentile	3	< 1	10	2	2
	Median	4	< 1	10	2	2
	75th Percentile	5	2	10	4	6
	Maximum	6	3	100	40	32
	No. of Obs.	24	23	10	25	22
	No. < D.L.	1	19	7	8	3
Duncan	Minimum	1	< 1	< 1	< 1	< 1
	25th Percentile	3	< 1	< 1	2	< 1
	Median	4	< 1	< 1	3	< 1
	75th Percentile	4	1	10	6	4
	Maximum	10	3	20	110	19
	No. of Obs.	53	67	21	71	53
	No. < D.L.	0	40	8	3	29
North Bend	Minimum	< 1	< 1	< 1	< 1	< 1
	25th Percentile	4	< 1	< 5	4	< 1
	Median	6	< 2	15	6	2
	75th Percentile	8	2	20	11	3
	Maximum	10	6	70	40	8
	No. of Obs.	29	28	27	29	28
	No. < D.L.	0	17	12	1	8
Louisville	Minimum	1	< 1	< 1	< 1	< 1
	25th Percentile	4	1	< 10	3	< 1
	Median	6	1	< 10	5	< 2
	75th Percentile	7	2	20	6	5
	Maximum	12	20	50	20	80
	No. of Obs.	59	59	49	60	58
	No. < D.L.	0	42	29	2	32

Table 45. (continued)						
Location		Silver Ag, diss. (µg/L)	Aluminum Al, diss. (µg/L)	Lithium Li, diss. (µg/L)	Selenium Se, diss. (µg/L)	Zinc Zn, diss. (µg/L)
Brady	Minimum	—	—	—	—	20
	25th Percentile	—	—	—	—	—
	Median	—	—	—	—	20
	75th Percentile	—	—	—	—	—
	Maximum	—	—	—	—	20
	No. of Obs.	0	0	0	0	1
	No. < D.L.	0	0	0	0	0
Overton	Minimum	< 1	< 1	10	< 1	< 1
	25th Percentile	< 1	25	15	2	6
	Median	< 1	50	20	2	8
	75th Percentile	< 1	75	25	2	20
	Maximum	1	100	30	3	930
	No. of Obs.	13	2	2	13	15
	No. < D.L.	6	1	0	1	2
Grand Island	Minimum	< 1	< 1	—	< 1	< 1
	25th Percentile	< 1	10	—	1	7
	Median	< 1	20	—	2	20
	75th Percentile	< 1	30	—	3	20
	Maximum	2	100	—	8	80
	No. of Obs.	23	58	0	23	25
	No. < D.L.	19	11	0	1	8
Duncan	Minimum	< 1	< 1	18	< 1	< 1
	25th Percentile	1	< 10	30	2	3
	Median	1	10	34	2	8
	75th Percentile	1	20	37	2	20
	Maximum	2	200	53	4	42
	No. of Obs.	65	48	47	65	71
	No. < D.L.	54	18	0	3	9
North Bend	Minimum	< 1	—	3	< 1	< 1
	25th Percentile	< 1	—	—	1	3
	Median	< 1	—	3	1	10
	75th Percentile	< 1	—	—	2	20
	Maximum	2	—	3	6	340
	No. of Obs.	25	0	1	28	29
	No. < D.L.	19	0	1	1	12
Louisville	Minimum	< 1	< 10	13	< 1	< 1
	25th Percentile	< 1	< 10	21	1	4
	Median	< 1	20	24	2	9
	75th Percentile	< 1	30	28	2	17
	Maximum	4	610	51	5	35
	No. of Obs.	63	47	47	70	59
	No. < D.L.	53	12	0	3	16

Table 46. Summary of organic contaminants (commonly used herbicides) at 6 sites in the mainstem of the Platte River in the Central and Lower Platte basins based on the period of record shown in Table 44					
Location		Simazine Diss. Water Rec. (µg/L)	Cyanazine Diss. Water Rec. (µg/L)	Atrazine Diss. (ppb)	2,4-D Whole Sample (µg/L)
Brady	Minimum	< 0.05	< 0.05	0.07	< 0.01
	25th Percentile	< 0.05	< 0.05	0.09	< 0.01
	Median	< 0.05	< 0.05	0.10	< 0.01
	75th Percentile	< 0.05	0.05	0.12	< 0.01
	Maximum	< 0.05	0.05	0.13	< 0.01
	No. of Obs.	2	2	2	6
	No. < D.L.	2	1	0	6
Overton	Minimum	—	—	0.37	—
	25th Percentile	—	—	- - -	—
	Median	—	—	0.37	—
	75th Percentile	—	—	- - -	—
	Maximum	—	—	0.37	—
	No. of Obs.	0	0	1	0
	No. < D.L.	0	0	0	0
Grand Island	Minimum	< 0.05	< 0.05	0.17	—
	25th Percentile	< 0.05	< 0.05	0.22	—
	Median	< 0.05	< 0.05	0.58	—
	75th Percentile	< 0.05	0.06	0.80	—
	Maximum	0.09	2.28	4.76	—
	No. of Obs.	24	24	24	0
	No. < D.L.	19	18	0	0
Duncan	Minimum	< 0.05	< 0.05	0.23	< 0.01
	25th Percentile	< 0.05	< 0.05	1.72	< 0.01
	Median	0.06	0.09	3.71	< 0.01
	75th Percentile	0.14	0.14	6.99	< 0.01
	Maximum	0.24	0.41	19.03	< 0.01
	No. of Obs.	15	15	15	4
	No. < D.L.	6	6	0	4
North Bend	Minimum	—	—	—	< 0.01
	25th Percentile	—	—	—	0.02
	Median	—	—	—	0.07
	75th Percentile	—	—	—	0.07
	Maximum	—	—	—	0.12
	No. of Obs.	0	0	0	10
	No. < D.L.	0	0	0	1
Louisville	Minimum	< 0.005	< 0.01	< 0.05	< 0.01
	25th Percentile	0.026	0.05	0.18	< 0.01
	Median	0.041	0.41	0.76	< 0.01
	75th Percentile	0.050	2.65	3.00	< 0.01
	Maximum	0.230	30.00	30.00	0.80
	No. of Obs.	91	91	162	14
	No. < D.L.	29	12	8	11

There are three different arsenic aquatic life criteria in Nebraska. The most restrictive is the 1.4 µg/L aquatic life criterion based on the protection of human health; the standard is meant to assure that bioaccumulation to concentrations that are associated with an estimated 10^{-5} (1 in 100,000) probability of cancer in human consumers of fish does not occur. The criterion has nothing to do with the health of the fish themselves.

The arsenic chronic aquatic life standards are 190 µg/L for As^{III} (arsenite) and 48µg/L for As^{V} (arsenate). These are used as a basis for comparison because the monitoring data are not speciated. However, the maximum concentration of dissolved arsenic is 30 µg/L; this would be the total of the As^{III}, As^{V}, and As^{-3} (arsenide), although the presence of large percentages of all three forms at once is chemically unlikely. Nevertheless, it is obvious that the chronic standard for the individual arsenic species have not been exceeded by any of the monitoring results. There does not appear to be any risk to aquatic life due to arsenic contamination.

Lead exceeds its standard in more than ½ of the samples at both Overton and Grand Island. However, there are a very limited number of samples, *i.e.* < 10 at Overton and < 20 at Grand Island (Table 44). The highest median lead concentration is at Overton at 4 µg/L. The median lead concentration is below the detection limit of 1 µg/L at Duncan and the detection limit of 2 at Louisville (Table 44). The calculated standards based on hardness range from < 0.1 to 1 µg/L. The exceedences of the standard at sites with the largest number of lead samples, *i.e.* Duncan and Louisville, are less than 50 percent, but still represent a significant percentage of the total. Based on the above, there appears to be a significant problem of lead contamination in the Platte River from Overton to its mouth. However, a closer inspection of the data indicates that there may really be no problem at all, at least under current conditions. Samples collected during the last several years of the record, the last of which was in 1991, were all below the detection limit. Although the detection limit may be well above the standard, the result would be assumed to meet the standard. There is no way to tell if it did not. The recent data would seem to indicate that the standard is met. A single exceedence would not be a violation. The standards are written such that a single exceedence may not actually violate a standard. The standards are evaluated by counting the percent exceedence in a set of samples over a specified period.

Figure 138 shows a plot of the lead data against the maximum lead standard at the Duncan gage (used as an example). The problem with lead may extend farther upstream, but there are not enough data between Overton and the North and South Platte confluence to make a valid assessment. For example, only one sample has been collected at Brady (Table 44). Figure 138 also provides and example of the changing detection limits mentioned earlier (note samples with "<" symbol).

Aluminum, cadmium, and silver have exceeded their respective standards in a small percentage of samples. Silver exceeded its standard in one sample each at North Bend and Louisville. Aluminum exceeded its standard in 2 samples each at Duncan and Louisville; however, the aluminum standard was exceeded in 20 percent of the samples collected at Grand Island. Based on this, aluminum may be a concern in the habitat reach.

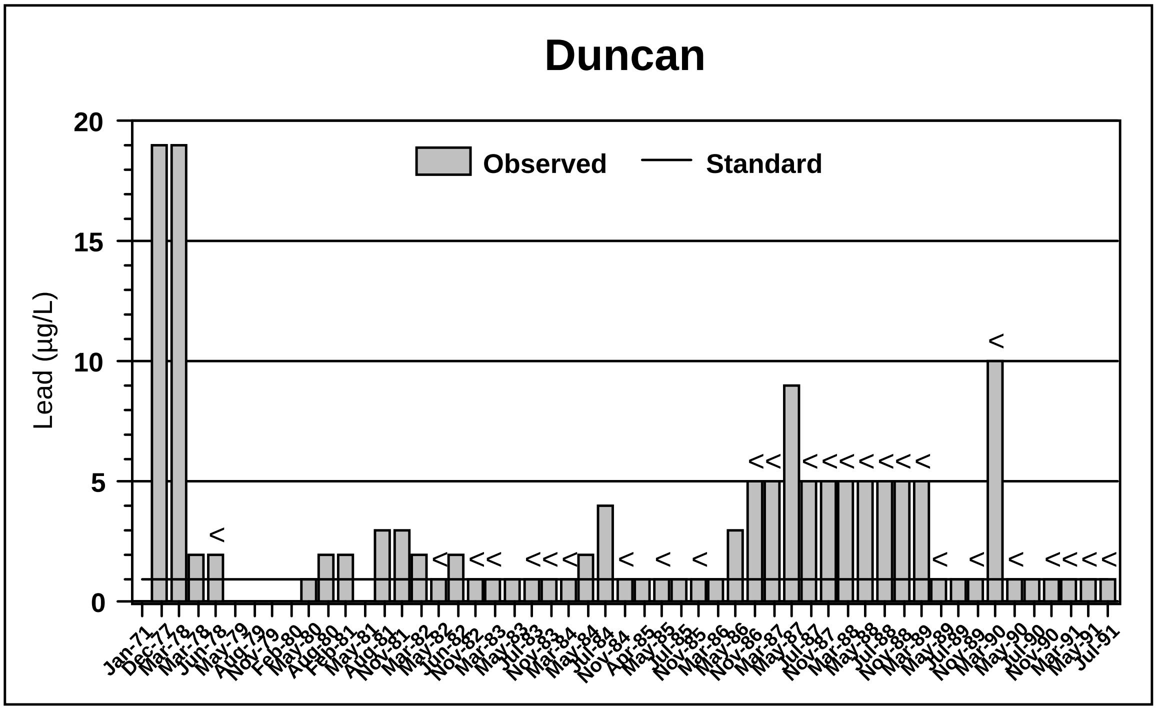


Figure 138. Comparison between lead concentrations and the maximum lead standard

Mercury is not included in the previous tables. Mercury presents a special case in that the standard is well below the detection limit that is usually obtained by the analytical techniques that are most often used. The current standard analytical technique for mercury is by cold vapor atomic absorption spectrophotometry, which has a detection limit of 0.1 (or sometimes as high as 0.5) µg/L. The mercury standard, based on the chronic warm water aquatic life criterion is 0.012 µg/L or about ⅛ the minimum detection limit. (The mercury standard is so low because it is based on the very high bioaccumulation potential of mercury.) In evaluating the data against the standard, it is assumed that any datum that is less than the detection limit is also less than the standard. This obviously may or may not be true; however, there is no way to determine otherwise. Alternatively a comparison to the detection limit will also suffice as a comparison to the standard, *i.e.* any sample that exceeds the detection limit will exceed the standard.

As can be seen in the data summary in Table 47, the available data consist of dissolved or total mercury. There are also a few suspended mercury analyses, but the most recent of these date from 20 years ago and are not being used. The mercury standard is for total recoverable mercury. The total recoverable analysis is based on a dilute acid digestion of the sample that is meant to break down particles that would be digested by an aquatic organism. It is meant to simulate biologically available mercury. The total mercury shown in Table 47 is from a strong acid digestion and would be expected to be somewhat higher than what would be obtained from the dilute acid digestion. However, in practice the total recoverable concentration is near the dissolved concentration. In this evaluation, both the dissolved and total mercury data will be evaluated against the standard.

Table 47. Dissolved and total mercury at 5 sites in the Platte River basin

A. Dissolved Mercury

	Overton	Grand Island	Duncan	North Bend	Louisville
Median (µg/L)	< 0.1	< 0.5	< 0.1	< 0.1	< 0.1
Maximum (µg/L)	0.7	< 0.5	0.4	0.5	1.0
No. of Obs.	11	23	54	29	58
No. > Std.	4	0	10	4	12
No. < D.L.	7	23	44	25	46
Begin Year	1978	1973	1971	1973	1974
End Year	1981	1981	1991	1981	1991

B. Total Mercury

	Overton	Grand Island	Duncan	North Bend	Louisville
Median (µg/L)	< 0.1	< 0.1	0.1	0.3	< 0.1
Maximum (µg/L)	6.0	0.1	1.2	0.7	1.5
No. of Obs.	38	28	18	31	46
No. > Std.	20	6	13	13	21
No. < D.L.	18	22	5	18	25
Begin Year	1978	1973	1977	1973	1974
End Year	1989	1984	1982	1984	1989

The comparison of the dissolved mercury data to the standard indicates that there are a considerable number of samples that have exceeded the standard (Table 47A). The greatest on a percentage basis is at Overton, but that gage has the fewest samples of the sites shown. None of the samples from Grand Island exceeded the standard, but the detection limit was always at 0.5 µg/L. Consequently it seems likely that some of the samples would have exceeded 0.1 µg/L. At the 3 sites downstream from the habitat reach, roughly 20 percent of the samples have exceeded the standard.

Another problem with the mercury data for the Platte Basin is that there is little in the way of mercury samples in recent years. The most recent data are from the Duncan and Louisville gages consist of dissolved mercury samples collected in 1991 (Table 47A). The most recent sample that exceeded the 0.1 µg/L detection limit was in 1988 at Duncan and in 1989 at Louisville.

A much greater percentage of the total mercury samples have exceeded the standard than was true of the dissolved fraction alone, based on the comparison to the detection limit (Table 5B). This is to be expected. This indicates that there was additional mercury present when the samples were collected, but there is no way to tell how much of it was biologically available. However, based on the biological data presented elsewhere, mercury is a problem in the Platte River.

The State of Nebraska DEQ (NDEQ, 1996c; 1998) listed pesticides as a major cause of nonsupport for beneficial uses in the Platte River mainstem. Table 32 indicates that atrazine had exceeded its standard a large percentage of the time, particularly in the lower reaches of the river. Figures 139 and 140 show histograms of atrazine monitoring data at 3 sites in the lower Platte mainstem in comparison to the water quality standard. These data, unlike the lead data, show definitively the frequency with which the standard is exceeded. The data for Grand Island show that the standard is exceeded occasionally, but not by a great amount (Figure 139). The maximum concentration is less than 5 µg/L. There are fewer data at the Duncan gage than at Grand Island (Table 46), but the percentage of samples that exceeds the standard is much larger (87 and 21 percent respectively), with 33 percent of the Duncan samples greater than 5 µg/L.

There are many more atrazine samples from the Louisville gage (162) than from either of the upstream gages (Table 46). Of the 162 samples from the Louisville gage, 69 exceed the 1 µg/L standard. This translates to 43 percent, which is about ½ the percentage at the Duncan gage. From this perspective it would seem that there is some improvement in water quality relative to atrazine contamination in the river at the farthest downstream station. Alternatively, although the frequency of exceeding the standard decreases, the magnitude of the peak concentration of atrazine increased (compare figures 139 and 140). The maximum atrazine concentration has been 19 µg/L at Duncan, but it has been as high as 30 µg/L at the Louisville gage.

The NDEQ has monitored herbicide concentrations in the Middle Platte Basin for their 303(d) analysis. The data from the NDEQ monitoring in 2002 and 2003 are summarized in Table 48. Although some of the atrazine concentrations are still comparatively high, none of them exceed the current aquatic life criteria for atrazine (compare the maximum with Table 49).

Table 48. Summary of NDEQ herbicide monitoring data from the Middle and Lower Platte Basins from 2002 and 2003

Site	Pesticide	Minimum	Median	Maximum	Samples	No. < D.L.[1]
Overton	Atrazine	< 0.05	0.135	0.64	24	4
	Alachlor	< 0.05	< 0.05	0.17	24	16
	Metolachlor	< 0.05	0.07	0.35	24	11
Grand Island	Atrazine	< 0.05	0.225	3.35	12	1
	Alachlor	< 0.05	< 0.05	4.4	12	9
	Metolachlor	< 0.05	0.08	1.26	12	4
Duncan	Atrazine	0.11	0.265	7.7	16	0
	Alachlor	< 0.05	< 0.05	0.64	16	11
	Metolachlor	< 0.05	< 0.05	1.16	16	9
North Bend	Atrazine	< 0.05	0.165	3.77	18	6
	Alachlor	< 0.05	< 0.05	0.16	18	14
	Metolachlor	< 0.05	0.06	0.95	18	9
Louisville	Atrazine	< 0.05	0.095	21.12	12	2
	Alachlor	< 0.05	< 0.05	2.28	12	8
	Metolachlor	< 0.05	0.1	10.67	12	5

[1] D.L. – Detection Limit – actually the reporting limit or limit of quantitation

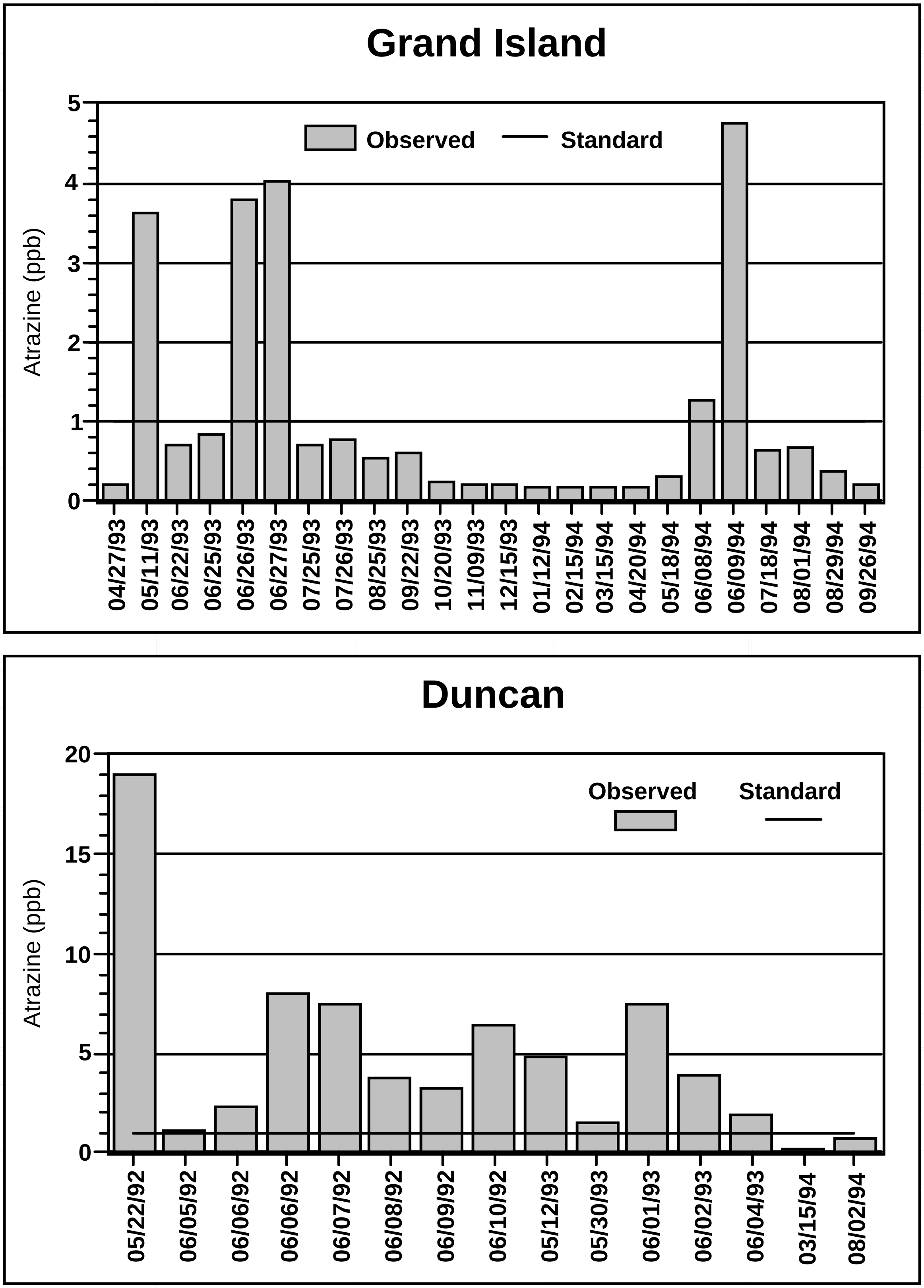


Figure 139. Comparison of atrazine concentrations in the Platte River at the Grand Island and Duncan gages and the atrazine aquatic life standard

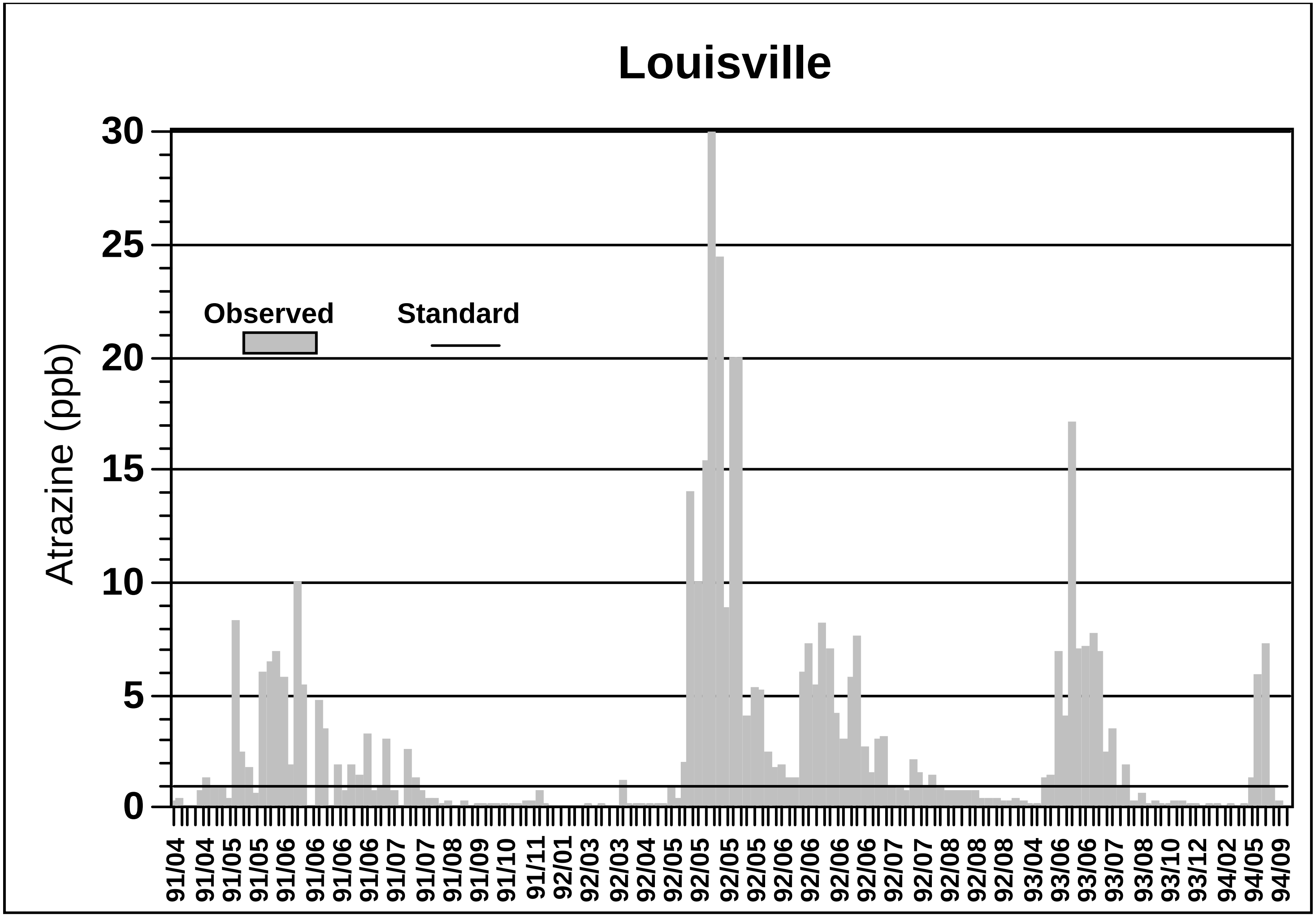


Figure 140: Comparison of atrazine concentrations in the Platte River at the Louisville gage and the atrazine aquatic life standard

Table 49. Nebraska Pollutant Criteria for 303(d) evaluations (all in µg/L)			
Pesticides	Aquatic Life		Public Supply
	Acute	Chronic	
Alachlor	760	76	2
Atrazine	330	12	3
Metolachlor	390	100	40

The preceding discussion of atrazine was what was used in the DEIS. At the time that it was prepared, the atrazine standard was based on the drinking water standard, but was assigned to aquatic life as well. The current atrazine standard is shown in Table 49, which also shows the criteria for alachlor and metolachlor. The public water supply criteria are applied to drinking water sources. The Platte River is not currently classified as a public water supply source and those criteria do not apply. The reason for including the public water supply criteria is for comparison with what was used for standards in the preceding discussion, when the drinking water standard of 1 µg/L was used as a basis for comparison. The stream segments in the Middle and Lower Platte Basins are classified for aquatic life support. Therefore, the herbicide standards would be based on the chronic aquatic life criteria. A comparison of the respective herbicide maximum concentrations in Table 48 with their aquatic life criteria shows that none of the criteria are exceeded. Consequently, the Middle and Lower Platte Basins are no longer listed for herbicide impairment of the aquatic life beneficial use by the NDEQ.

The median herbicide concentration should be representative of the trends with distance downstream in the Platte River. The median atrazine concentration increased from Overton to Grand Island to Duncan (Table 48). The trend shown by the median atrazine concentration is mirrored by the maximum concentrations. Duncan is also the only site at which the minimum atrazine concentration is above its detection limit. Downstream from Duncan, atrazine concentrations show a decrease to North Bend to Louisville, a trend not shown by the maximum concentrations. Although the maximum atrazine concentration decreases from Duncan to North Bend, the maximum at Louisville is much higher than any concentration anywhere else in the basin. The very high maximum atrazine concentration at Louisville is consistent with some of annual peaks shown above on Figure 140.

Alachlor shows no trends with distance downstream. The median alachlor concentration is below its detection limit at all sites (Table 48). However, alachlor does show an anomalously large maximum concentration at Louisville.

The median metolachlor concentration shows no particular trend with distance downstream either. However, the lowest median, which is below the detection limit, was observed at the Duncan gage (Table 48). Metolachlor does show the same type of large maximum at Louisville as was shown by atrazine and alachlor.

Temperature Standard

There have been a number of fish kills over the years in the Platte River mainstem (NGPC, 1996). Ninety-two percent of these were in the Middle Platte between Cozad and Columbus (*ibid.*). Many of these fish kills were attributed to low flows and excessive temperatures. These fish kills included species that are known to provide forage for nesting least terns. For this reason the temperature issue is a significant concern for the recovery of the tern in the Middle Platte Basin.

Between 1988 and 1995, the Fish and Wildlife Service monitored temperature at up to 5 sites in the Platte River. The data and a comparison to the Nebraska water quality standard for temperature in the Platte River (32°C or 90°F) for each year and each site are presented in Table 50. The number of days shown in Table 50 represents the total between the initiation of monitoring (June 1) and the last day of the monitoring period. The last day during the years 1988 through 1993 was August 31. The end of the monitoring period was extended to the first or second week in September in 1994 and 1995; so the number of days in the monitoring period increased in those years. The number of observations represents the number of days within the total that monitoring actually occurred. This latter has the potential to affect the results, in that the days when the monitors were not operating could be ones on which the standard was exceeded. For this reason, the percentage shown in the last column, which is based on the number of days on which measurements were made, is a more representative measure of the exceedence of the standard than the number of days per year on which the standard is exceeded.

Table 51 shows annualized values for the exceedences of the temperature standard in the Platte River. The data in Table 51 indicate that the exceedences of the standard increase in a downstream direction. They are the most frequent at the lower end of the critical habitat reach of the Platte River.

There is some controversy over the cause and best way to control temperature in the Middle Platte. Dinan (1992) simulated temperatures in the Platte River and found that the frequency and duration of lethal temperatures for fish could be reduced if flows were increased. This study was critiqued by Miller (1994) who maintained that the air temperature was the major factor in bringing about excessive water temperature and developed a series of regression relationships to evaluate the relationships between air temperature, water temperature, and flow. Zander (1996) also evaluated the relationships using several statistical techniques and reaffirmed Dinan's earlier findings. At about the same time Miller (1996), following a further evaluation of the air and water temperature and flow interrelationships, reiterated that it was extremely unlikely to change water temperature with flow manipulations alone.

The problem with all of this is that the there is no real difference in the findings. Neither Dinan (1992) nor Zander (1996) was recommending flows for temperature control. In both cases they found that any increase in flow would result in less warming of the water. Consequently, at the higher summer flows anticipated as a result of the preferred alternative from the ESA consultation on the FERC licensing of the Kingsley hydro plant mentioned above in the Lake

Table 50. Summary of Temperature Data and Comparison to the Nebraska Temperature Standard at 5 Sites in the Platte River					
Site	Year	No. of Days	No. of Obs.	No. > Std.	% > Std.
Overton	1988	92	0	0	No Data
	1989	92	20	0	0.0%
	1990	92	77	3	3.9%
Ave. %	1991	92	88	1	1.1%
> Std. =	1992	92	85	0	0.0%
2.8%	1993	92	82	1	1.2%
	1994	104	91	12	13.2%
	1995	101	35	0	0.0%
Odessa	1988	92	0	0	No Data
	1988	92	83	13	15.7%
	1990	92	63	12	19.0%
Ave. %	1991	92	75	12	16.0%
> Std. =	1992	92	69	1	1.4%
9.5%	1993	92	85	0	0.0%
	1994	104	91	10	11.0%
	1995	101	32	1	3.1%
Kearney	1988	92	0	0	No Data
	1989	92	38	6	15.8%
	1990	92	86	30	34.9%
Ave. %	1991	92	92	36	39.1%
> Std. =	1992	92	68	5	7.4%
19.2%	1993	92	85	4	4.7%
	1994	104	90	29	32.2%
	1995	99	4	0	0.0%
Mormon	1988	92	43	28	65.1%
Island	1989	92	0	0	No Data
	1990	92	65	42	64.6%
Ave. %	1991	92	90	53	58.9%
> Std. =	1992	92	70	6	8.6%
37.5%	1993	92	85	16	18.8%
	1994	93	74	24	32.4%
	1995	93	71	10	14.1%
Phillips	1988	92	0	0	No Data
	1989	92	81	39	48.1%
	1990	92	82	50	61.0%
Ave. %	1991	92	69	48	69.6%
> Std. =	1992	92	56	12	21.4%
34.5%	1993	92	84	9	10.7%
	1994	104	92	28	30.4%
	1995	101	6	0	0.0%

Ogallala section of this report, there should be less frequent exceedences of the temperature standard as a side benefit.

The FWS (1997) in the Biological Opinion for Kingsley Hydroelectric Relicensing relied on a hydrodynamic temperature model (Sinokrot *et al.*, 1997) to evaluate the relationship between flow and temperature. The conclusion there was that the increase in the frequency of flows

Table 51. Annual Average Exceedence of the Temperature Standard at 5 Sites in the Critical Habitat Reach of the Platte River				
Site	No. of Days	No. of Obs.	No. > Std.	% > Std.
Overton	95	68	2	3.6%
Odessa	95	71	7	9.8%
Kearney	95	66	16	23.8%
Mormon Island	92	71	26	35.9%
Phillips	95	67	27	39.6%

above 1,200 ft^3/s if the preferred alternative were implemented would decrease the frequency of exceeding the temperature standard at Grand Island significantly and improve habitat conditions for the fish community.

Zander (1996) looked at the probability of exceeding the 32°C temperature standard at various flows. That approach has been taken here as well. Using the temperature data for the 2 sites at which the standard was consistently exceeded in the 1988-95 data sets, a series of cumulative frequency distributions were developed for various increments of flow at the USGS Grand Island gage (Figure 141). The flow increments were sized to include a minimum of 25 observations in each frequency distribution. This leads to an unequal distribution of flows across each interval. The flow intervals were also determined to some extent by the goal of maintaining at least 12 frequency intervals. The probability of exceeding 32°C at the Mormon Island and Phillips sites at different flows are shown on Figure 142. The coefficients of determination (R^2) of the regressions of the probability of exceeding the temperature standard as a function of flow is also shown on each of the plots in Figure 142; the regressions explain 80 percent or more of the variation in the probability - flow distributions. The regressions represent a polynomial (actually a quadratic) fit of the probabilities to the flow at the center of each interval.

The probability - flow distribution for the Mormon Island monitoring site indicates that the probability of exceeding the temperature standard during the months of June to September is greater than 60 percent at flows less than 100 ft^3/s, around 50 percent at flows between 100 and 500 ft^3/s, 30 percent at flows between 500 and 1500 ft^3/s, and so forth (Figure 142A). This does illustrate that there is a lower probability of exceeding the water quality standard at higher flows. However, the upper end of each of the probability distributions still show that at even the highest flows, there is a possibility of exceeding the temperature standard in the vicinity of Grand Island.

In the case of the Central Platte, the water quality indicator selection also involved a review of the Nebraska 303(d) list (1996). The list included the water quality standards that were being violated, *i.e.* pesticides, ammonia, and pathogens at that time. The current draft 303(d) lists includes pathogens in the Middle Platte mainstem as the basis for required TMDL's. The temperature standard is also listed as having not been met 30 and 27 percent of the time in the 2 Middle Platte reaches respectively in 2001 (NDEQ, 2002b). The water quality measures, along with TDS and EC, were then correlated with flow to evaluate potential relationships. This will be described further in the next section on Methods of Analysis.

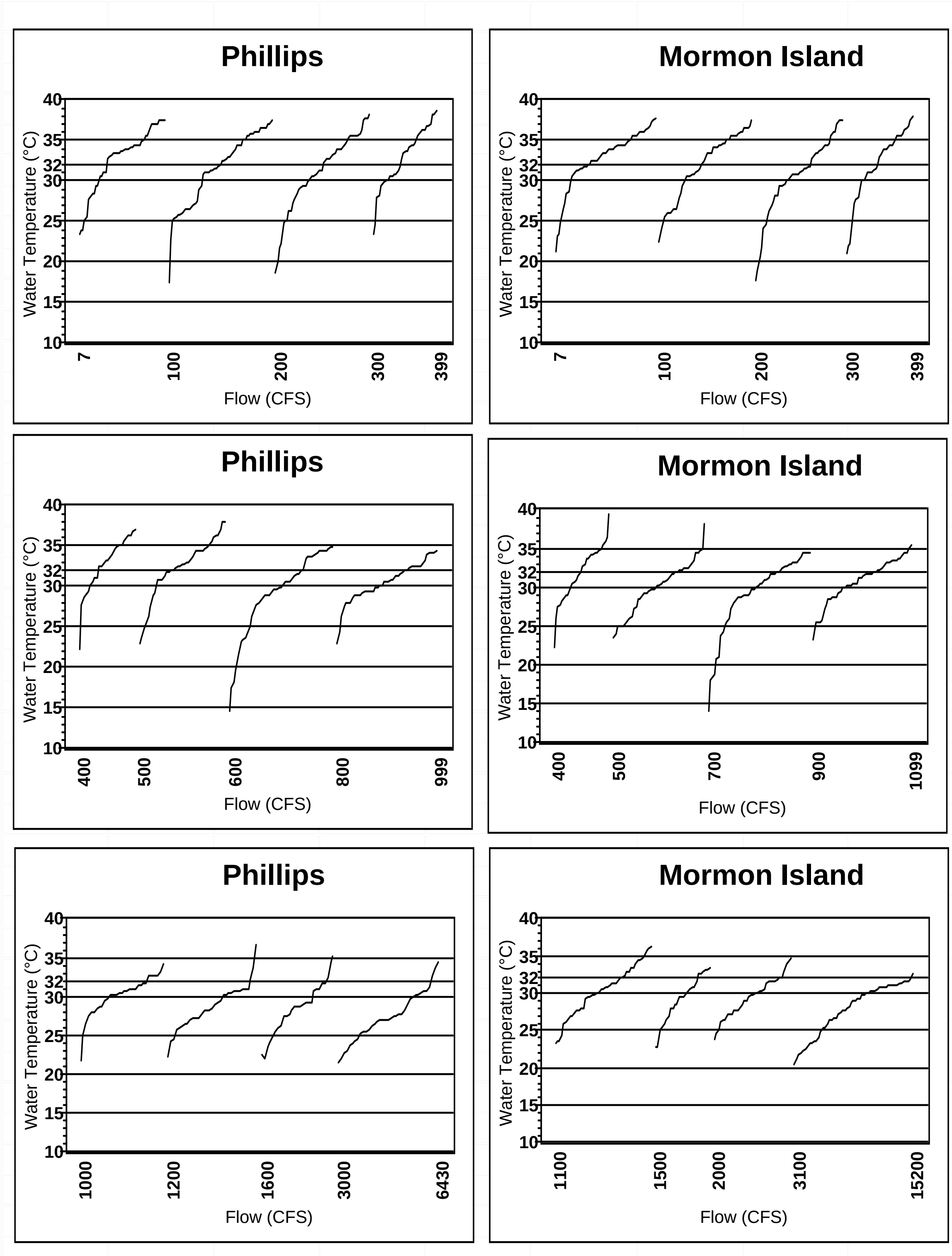


Figure 141. Cumulative frequency distributions of maximum daily water temperatures within various intervals of flow at the Grand Island gage with a comparison to the water quality standard for temperature

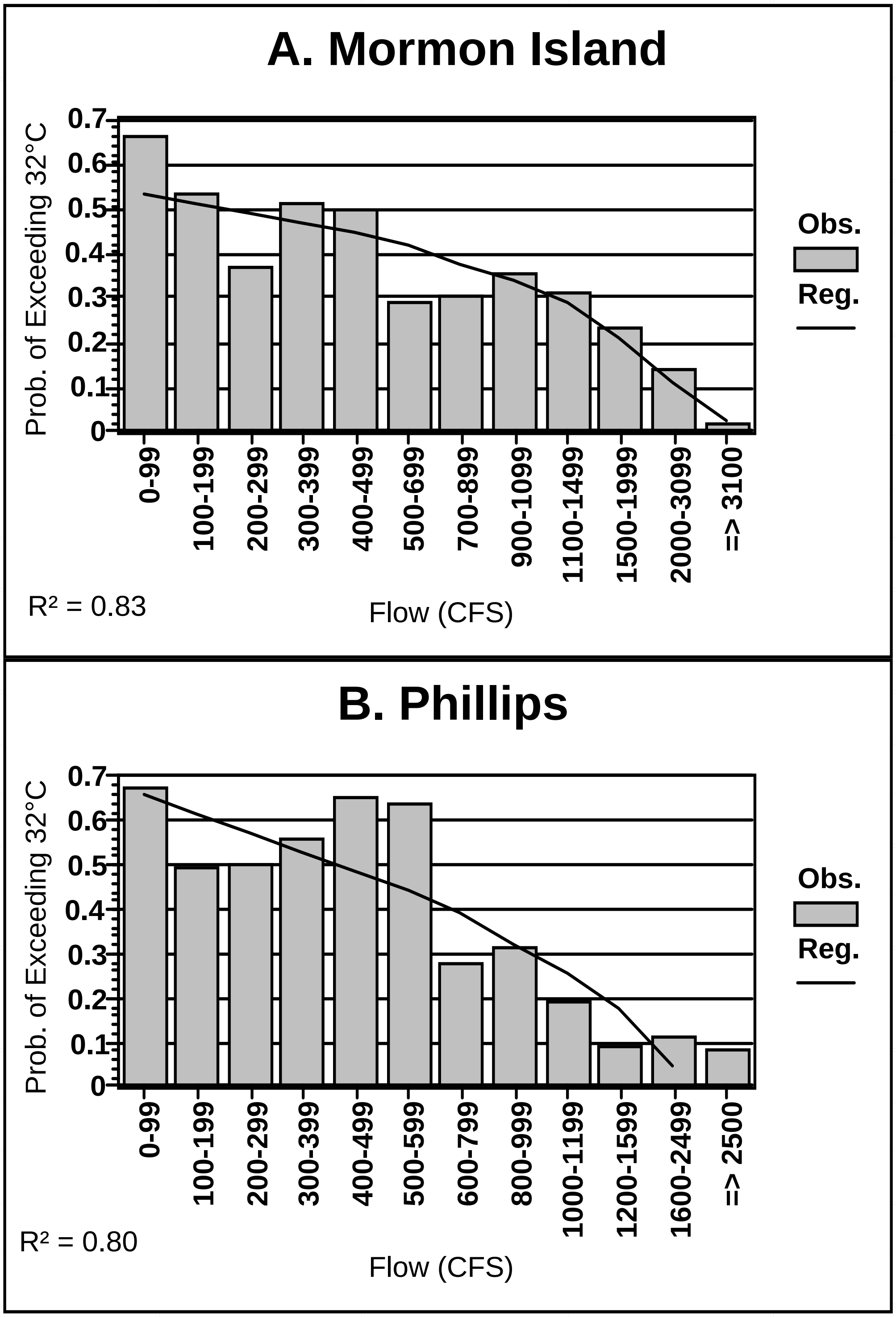


Figure 142. Flow *vs.* Probability Histograms for Platte River Temperatures

Another aspect of the Platte River Endangered Species Recovery Program is sediment manipulation. There is no water quality standard for suspended sediment, as usually measured by total suspended solids (TSS). However, turbidity is included in the standard for aesthetics and is also mentioned as a possible factor in biological toxic effects. Turbidity was selected as an indicator of the effects of sediment manipulation.

Other Flow Relationships

Another important criterion for selecting water quality indicators for the impact analysis was that there had to be a relationship between the potential indicator and flow. The main effect of the Platte River Endangered Species Recovery Program is to manipulate flows to improve migratory habitat for the whooping crane and nesting habitat for the interior least tern and the piping plover. If there is no relationship between a potential indicator and flow, then there should be no impact due to flow changes.

TDS (or specific conductance) was initially proposed as the principal indicator for the comparison of alternatives. As was illustrated in the North Platte Basin, there is frequently an inverse relationship between TDS (and/or specific conductance) and flow (c.f. Hem, 1985). However, as is shown in Table 52, there are no such relationships in the habitat reach (approximated by the reach between Overton and Grand Island), although there are significant relationships between TDS (and specific conductance) and flow in both the North and South Platte Rivers, as was noted above.

The absence of a relationship between TDS and flow is common below larger reservoirs, where the seasonal fluctuations in flow and TDS are damped by mixing in the reservoir pool, as was shown below Seminoe on the North Platte River in Wyoming. It also common in highly regulated systems, where TDS changes tend to be independent of flow because the regulatory system is the primary source of both the flow and the TDS. For the relationship between TDS and flow to be significant, there must be a high degree of dilution occurring. Both of the above factors, *i.e.* the presence of a large reservoir and a highly regulated hydrologic system, are at work in the Middle Platte.

Most of the alternatives being considered involve flow augmentation. Any indicator that is used to show effects of the alternatives must have some relationship to flow. Because the primary source of augmentation water in most of the alternatives is Lake McConaughy, which is the source of much of the flow now, the TDS should be similar to what it is now with implementation of any of those alternatives.

Pathogens, as represented by fecal coliform bacteria, are listed as a water quality problem in all of the reaches in the Middle and Lower Platte basins by the NDEQ (1998 - Table 53; NDEQ, 2002a; 2004a). As is discussed in the later section on Impaired Waters, there is a TMDL for bacteria in the Middle Platte Basin (NDEQ, 2003b). Regressions of fecal coliforms on flow are summarized in Table 52. There is a significant relationship between the bacterial concentration and flow in the South Platte near Julesburg and in the North Platte at the State Line. There is no significant relationship between fecal coliforms and flow in the habitat reach.

Table 52. Summary of relationships to flow for 4 Platte River gage sites									
Gage	Dependent Variable	Units	Geometric Mean	Geo. Mean Flow [cfs]	No. of Obs.	r^2	r	Slope	Constant
Overton	Sp. Cond.	µmho/cm	897	1742	209	0.008	0.092	0.01081	6.724610
	TDS	mg/L	587	784	95	0.004	-0.063	-0.00941	6.438182
	Fecal Coli.	#/100mL	176	2651	128	0.011	-0.103	-0.20170	6.874646
	Atrazine	mg/L	0.37	NA	1	Not enough data			
	NH_3-N	mg/L	0.0017	2290	144	0.001	-0.038	-0.03992	-6.092394
	Turbidity	JTU	13	817	24	0.275	0.524**	0.36575	0.141829
Grand Island	Sp. Cond.	µmho/cm	892	2245	220	0.024	0.156*	0.01923	6.645669
	TDS	mg/L	586	1028	146	0.002	0.044	0.00499	6.338757
	Fecal Coli.	#/100mL	140	2947	154	0.009	-0.097	-0.14888	6.136220
	Atrazine	mg/L	0.56	1569	24	0.047	0.216	0.44613	-2.221547
	NH_3-N	JTU	0.0018	3294	213	0.001	0.027	0.03653	-6.752021
	Turbidity	mg/L	16	966	86	0.048	0.219*	0.13621	1.841913
Duncan Platte	Sp. Cond.	µmho/cm	817	2763	321	0.002	-0.049	-0.00668	6.764501
	TDS	mg/L	550	2594	260	0.003	0.055	0.00823	6.250390
	Fecal Coli.	#/100mL	104	2372	101	0.002	0.050	0.09282	3.921627
	Atrazine	mg/L	3.01	1422	15	0.004	-0.059	-0.15070	2.195166
	NH_3-N	mg/L	0.0012	2444	135	0.005	-0.073	-0.09683	-6.044695
	Turbidity	JTU or NTU	15	1531	120	0.238	0.488**	0.53364	-1.370479
North Bend	Sp. Cond.	µmho/cm	481	6691	192	0.002	-0.046	-0.01503	6.295600
	TDS	mg/L	295	4304	100	0.109	-0.331**	-0.09124	6.451894
	Fecal Coli.	#/100mL	482	8157	149	0.019	0.139	0.30815	3.340162
	Atrazine	mg/L	No data		0	No data			
	NH_3-N	mg/L	0.0013	8527	180	0.000	-0.016	-0.02896	-6.344837
	Turbidity	JTU or NTU	40	3156	71	0.134	0.367**	0.33619	0.976580

* indicates that the r-value shown is statistically significant at the 0.05 α-level
** indicates that the r-value shown is statistically significant at the 0.01 α-level

Any significant regression between bacteria and flow is usually positive.   The relationship is usually due to an association with erosion and storm runoff. Alternatively, if a point source is contributing the contamination in a consistent manner, then the relationship would be negative. If the source is inconsistent, such as from erosion, but not tied to runoff, as in the case of a point source discharge from a treatment plant that has an occasional breakdown in disinfection, then there may or may not be a flow relationship. This last seems to be the case in the Middle Platte (Table 52). The bacteria TMDL addresses both point and nonpoint sources (NDEQ, 2003b).

Based on Table 53, another basis for noncompliance with water quality standards was related to pesticides. The pesticide that is of most concern in the Middle Platte is atrazine. Although atrazine is not currently listed, the Program should not affect atrazine concentration to the point that relisting is possible. At four of the seven sites shown in Table 52, there are not enough atrazine data to evaluate a relationship to flow. There is no relationship to flow at the other three sites, based on the USGS data. The 2002 NDEQ data do show a relationship between atrazine and flow at Louisville, but the site is not within the modeled reach of the Platte River and flow changes resulting from the Program alternatives at the gage cannot be quantified.

Atrazine is a highly significant contaminants problem in ground water in the Middle Platte basin. For the alternatives that involve ground water, atrazine could be considered. A much more definitive discussion of atrazine in ground water in Nebraska is provided in NDEQ (2004b). For surface water alternatives atrazine will be considered as it relates to ground water. For the most part, strictly surface water flow manipulation activities should not have an effect on atrazine concentrations in the river.

Ammonia was also listed as a basis for noncompliance with water quality standards in the Middle Platte basin (Table 53), but it has since been removed from the list at all sites in the Middle Platte Basin (NDEQ, 2004a). As was the case with the previous causes of noncompliance, there is no significant relationship between flow and ammonia in the Middle Platte basin (Table 52). Consequently ammonia, like the previously considered indicators, would not work well as an indicator.

The last potential water quality indicator shown in Table 52 is turbidity. There is a significant regression relationship between flow and turbidity at all 4 of the stations shown in Table 52. The poorest of the of the significant relationships is at the Grand Island gage; the turbidity-flow relationship for the Grand Island gage is illustrated on Figure 143. It could be noted that none of the r^2-values exceeds 0.5; so none of the relationships explains even half of the variation in turbidity. What this means is that the poorer regressions will show a poor response in the dependent variable to changes in the independent variable. In other words the sensitivity of the relationship to flow changes is somewhat less than ideal.

Table 53. Summary of water quality problems in the Middle Platte and Lower Platte reaches - Nebraska 1998 303(d) List

Waterbody ID	Waterbody Name	Size of Waterbody Affected	Specific Pollutant	Probable Pollutant Source	TMDL Status	Implementation of Water Quality Controls	Priority for TMDL (H,M,L,N/A)	Target for TMDL Next 2 Years?
MP1-20000	Platte River	53.6 mi.	200, 1700	200, 1000	Not Completed		L	No
MP2-10000	Platte River	60 mi.	600, 1700	200, 1000	Partially Completed[1]	Initiated	L	No
LP1-10000	Platte River	32.5 mi.	1700	200, 1000	Not Completed		L	No
LP1-20000	Platte River	66.5 mi.	1700	200, 1000, 4000	Not Completed		L	No

[1] Partially Completed means TMDL's (Total Maximum Daily Loads) have been calculated for a portion of the parameters, particularly those associated with point source discharges.

Specific Pollutant or Stressor:		Probable Source:	
200	Pesticides	200	Municipal Point Source
600	Ammonia (un-ionized)	1000	Agriculture
1700	Pathogens	4000	Urban Runoff/Storm Sewers

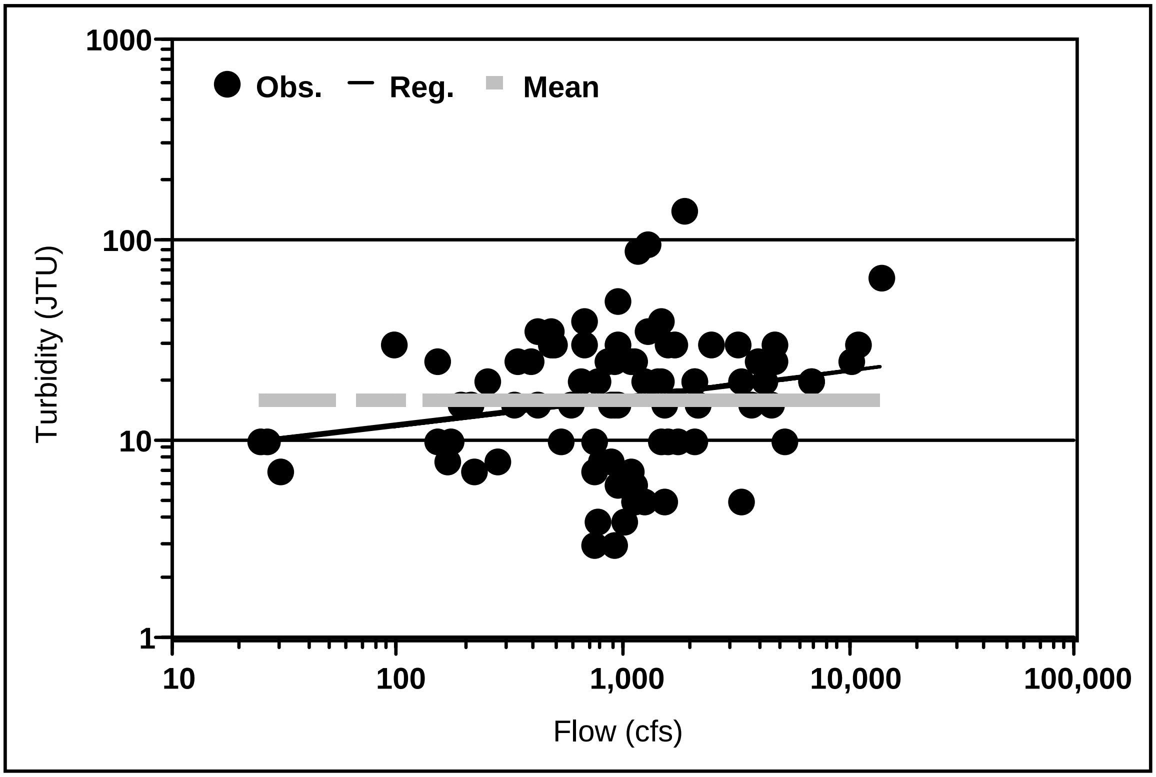


Figure 143. Scatter plot of turbidity as a function of flow at the Grand Island gage. Also shown are the predicted values from the turbidity-flow regression and the geometric mean turbidity at the gage for the period of record.

Despite the above concerns, turbidity could be a good indicator variable. One of the goals of the program is to manipulate sediment as well as flow. Suspended solids are a primary factor in controlling turbidity; however, the relationship between turbidity and total suspended solids (TSS) may be somewhat nebulous. TSS measures the concentration of the weight (or mass) of material suspended in the water at the time of sampling. The suspended material may consist of silt, clay, organic matter, plankton, and other microscopic organisms. Turbidity is a measure of the scattering of light as it passes through the water. Scattering is affected not only by the mass of TSS, but also the size and the shape of the particles. Greenberg *et al.* (1992) indicate that turbidity of water is caused by clay, silt, finely divided organic and inorganic matter, soluble colored organic compounds, and plankton and other microscopic organisms. For example, the relationship between color and turbidity is shown on Figure 144. The relationship between color and turbidity is better than the relationship between flow and turbidity based on their comparable r^2-values, *i.e.* 0.048 for flow and 0.264 for color. It should be noted that there is no significant relationship between flow and color. The multiple regression of turbidity on flow and color explains less of the variation in turbidity ($R^2 = 0.124$) than the one based on color alone ($r^2 = 0.264$ – Figure 144).

Turbidity is an expression of the optical property that causes light to be scattered and absorbed rather than transmitted in straight lines through the water. Correlation of turbidity with the weight concentration of suspended matter is difficult because the size, shape, and refractive index of the particulates affect the light-scattering properties of the suspension (*ibid.*).

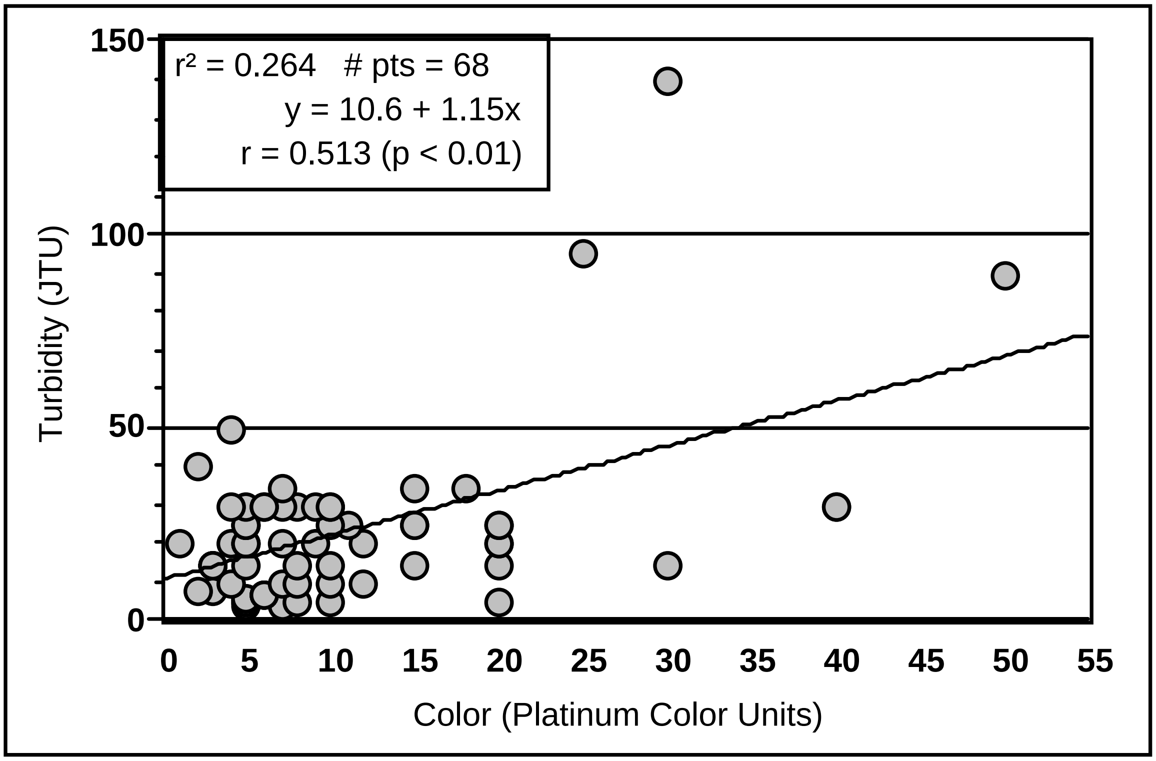


Figure 144. Regression relationship between turbidity and color at the Grand Island gage

Nevertheless, based on the list of factors causing turbidity, smaller particles will usually have a greater effect on turbidity than larger particles.

To further evaluate the above issue, the relationship between particle size and TSS was investigated. Most of the TSS observations also include the percent of the total that was in the <0.062 mm fraction. Particles that fall within this fraction are the silts and clays. As noted above, this is the fraction that is more likely to have a direct effect on turbidity. Plots of the two size fractions (<0.062 mm and $0.062 mm) against TSS are shown on Figure 145.

Figure 145A shows the relationship between the fine particles and TSS. The r^2 for a quadratic regression of the fine particles on TSS is 0.83, which is a very good fit to the data. What the relationship indicates is that the concentration of fines increases rather rapidly at lower concentrations of TSS, but levels off at higher concentrations. The curve is similar to that for the plot of the turbidity *versus* TSS shown on Figure 1. This seems to indicate that there is some physical validity to the relationship between turbidity and TSS over and above a coincidental relationship to flow.

Figure 145B shows the relationship between the coarser particles (sand-sized and greater) and TSS. Like that of the fine particles, there is also a quadratic fit of the coarse-grained data shown Figure 145A.

Where the fine-grained particles showed a lesser increase at higher concentrations of TSS, the coarse-grained particles increase at a higher rate at higher concentrations. This is a reflection of

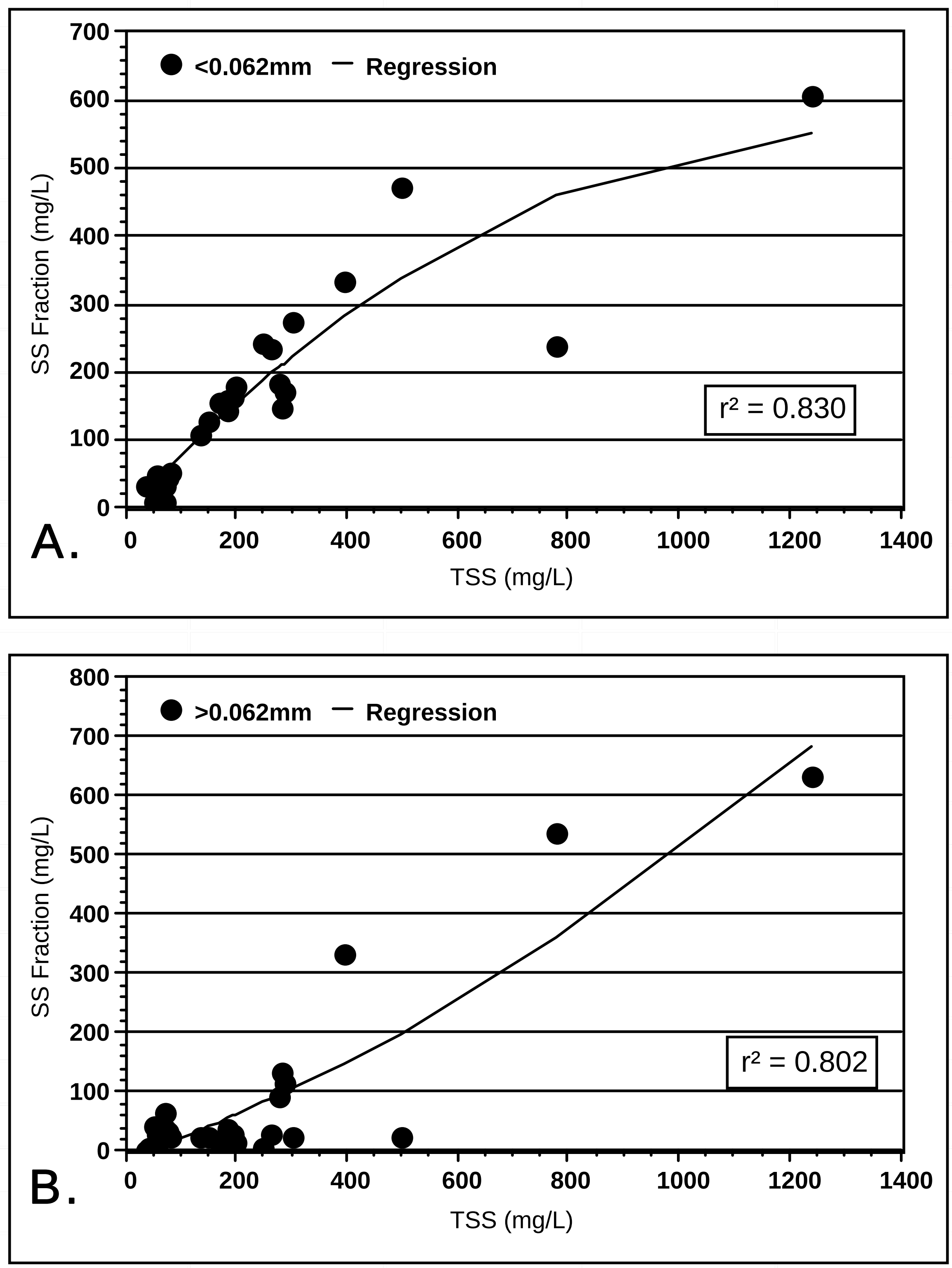


Figure 145. Relationships between <0.062 mm and 0.062 mm particle-size fractions and TSS at the Grand Island gage

the fact that higher concentrations of TSS are present at higher flows. The higher flows have a greater capacity to carry larger particles. On the other hand, these larger particles have little effect on turbidity.

Despite all of the above, turbidity will probably not serve as a good indicator of water quality effects. The alternatives that increase flow will increase TSS by increasing the erosion of bed materials. Figure 146 shows particle sizes for the bed material at the Overton and Grand Island gages during five months during 1998. None of the samples for the bed material contain measurable amounts of particles in the <0.062 mm size class. The predominant size class is the 0.5-1 mm fraction at the Overton gage and the 0.25-0.5 mm fraction at the Grand Island gage (Figure 146). These are three to four size classes coarser than the silt-clay fraction of interest. Since the size class of particles that is most responsible for causing turbidity is absent, there should be no observable effect on turbidity due to the increased erosion of bed material. As a consequence, all of the alternatives should have about the same effect; thus turbidity may not provide a good gage of effects for purposes of alternatives comparison.

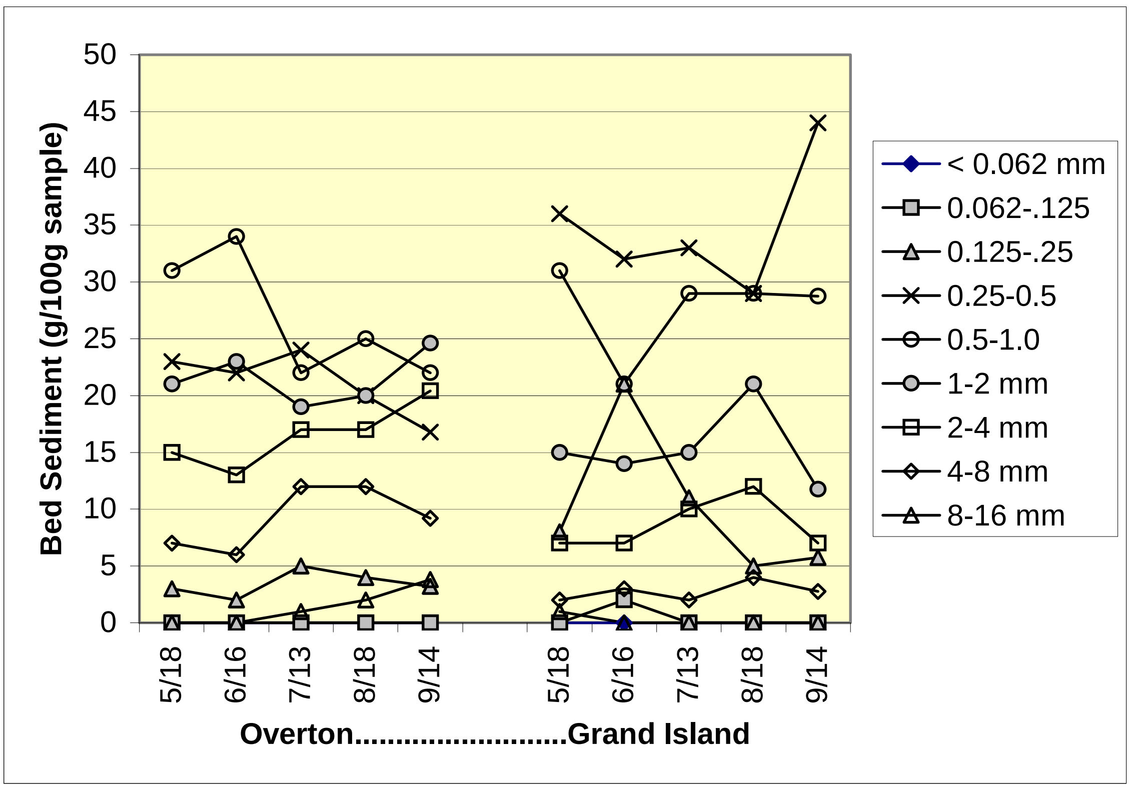


Figure 146. Particle size composition of bed material near the Overton and Grand Island gages on the Platte River

There is a management element that will increase sediment in the river due to erosion of islands. Samples of the bank sediments from the river and islands were collected in 2000. The difference in the distribution of sediment size fractions between the bed and the bank/island sediments is further illustrated on Figure 147. The figure was prepared by averaging the data for any site that had more than one sample. For example, there were 3 bank/island samples each from the sites at Lexington, Elm Creek, Kearney bridge, and Kearney. There were six samples from the sites at

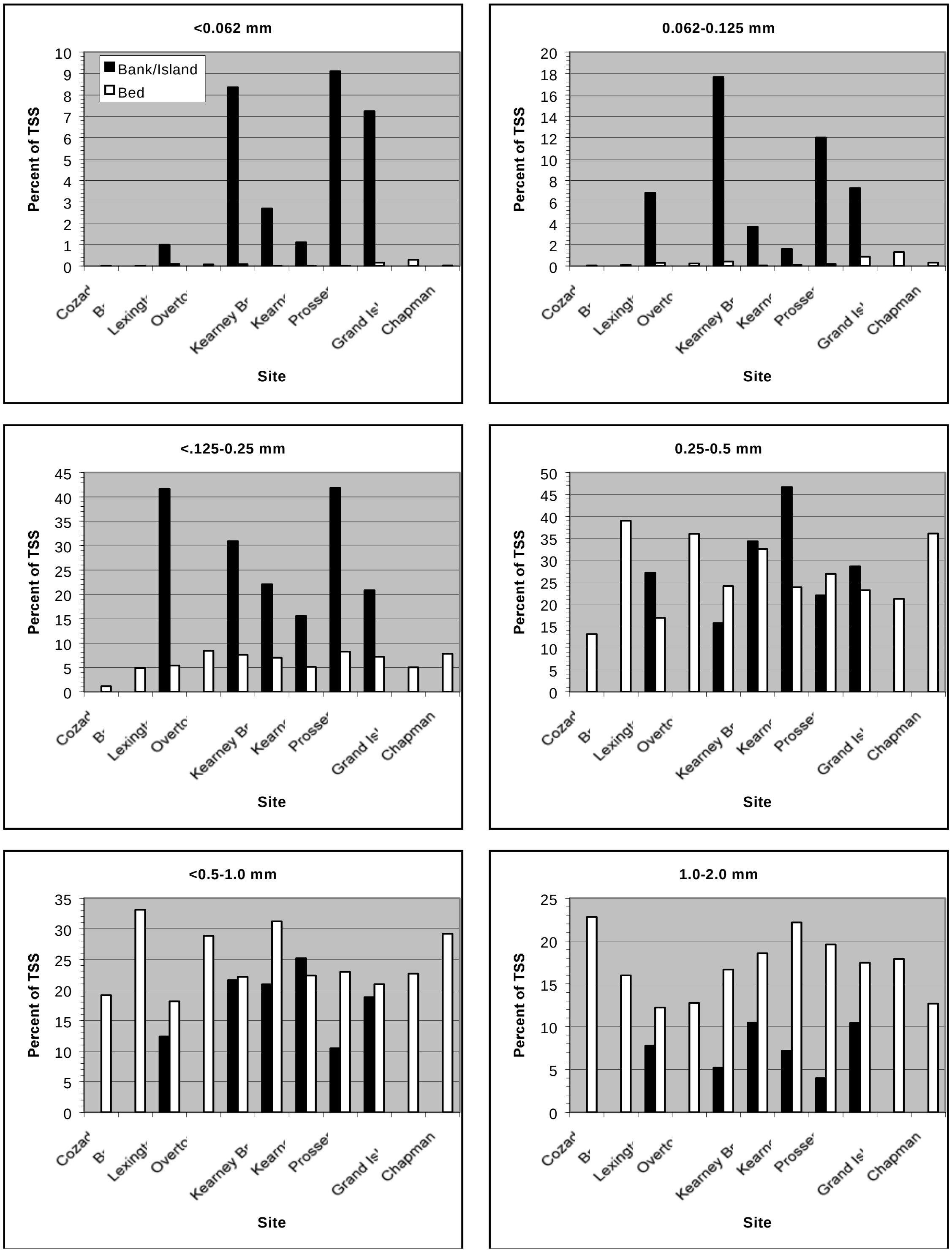


Figure 147. Comparison of grain size distributions for the bed and bank/island sediments by sample site

Alda and Prosser, 3 each at different depths from the banks and the islands.   With a couple of exceptions, the percentage of bed sediment in the <0.062 and the 0.062-0.125 size fractions are so small that they do not appear on their histograms on Figure 147. Although it may be difficult to discern from Figure 147, there are fine sediments present at 3 of the sites, with a maximum of nearly 10 percent at the Prosser site and more than 8 percent at the Elm Creek site. Consequently, there is a possibility of increasing the turbidity of the Platte River due to sediment manipulation. Turbidity will be used to reflect the water quality effects of sediment manipulation in the Central Platte River.

There are other differences in the grain size distributions between the bed and the bank/island samples. For example, in the 0.125-0.25 mm size fraction, the bed sediment samples appear on the histogram, but all are below the percentages shown by the bank/island samples. In the 0.25-0.5 mm size fraction, there is a mixed result in that there are several of the bank/island samples that are higher than several bed samples and *vice versa*. At the two larger fractions on Figure 147, the bed sediments have the higher percentages.

Environmental consequences

Methods of analysis

As was noted in the previous section, water temperature is only indirectly related to flow. One of the unique properties of water is that it has a high specific heat (the amount of heat required to raise the temperature of unit mass by unit amount). If the flow is high, there is a greater mass of water than at a lower flow. The temperature of the greater mass of water will rise more slowly than would that of a smaller mass of water under a given exposure to solar radiation.

In the previous section, temperature data from June, July, and August for the site at Mormon Island near Grand Island were used to develop frequency distributions of the water temperatures within different flow intervals. The temperatures within each of the flow intervals were compared to the temperature standard to calculate a frequency of exceeding the standard within each interval. Regression relationships based on polynomial fits were developed. On Figure 148, the frequencies of exceeding the standard are fitted to a Weibull model using nonlinear regression. The final curve and its formula are also shown on Figure 148. The r^2 for the fitted curve is 0.891, which means that the curve explains about 89 percent of the variation in the dependent variable, the probability of exceeding the water quality standard. As is indicated by the intercept of the curve, the upper limit of the probability of exceeding the standard that can be calculated from the curve is 0.86. In addition the highest bracket of flow on the *x*–axis is shown as "$3100" ft³/s. The actual value used in calculating the curve was the median of the flow data in that bracket, which is 6,497 ft³/s. For all of the other flow data points, the midpoint of the interval was used as the *x*–value for the curve.

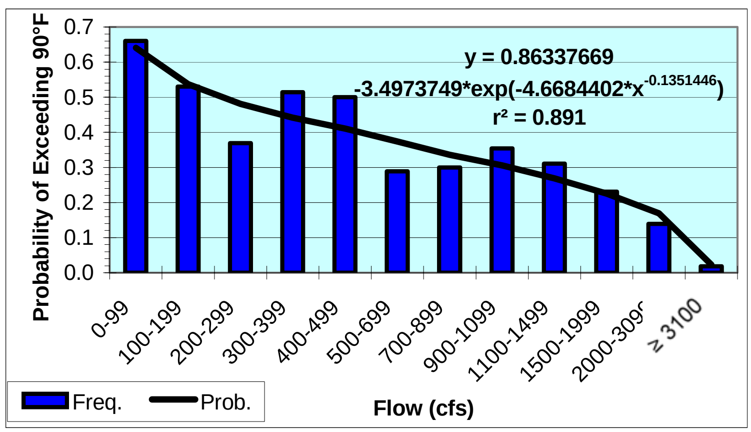


Figure 148. Frequency distribution and fitted curve for probability estimation for exceeding the temperature standard in relation to the flow at Grand Island

As was noted above, in the biological opinion for the FERC licensing of Kingsley Dam power plant, the U.S. Fish and Wildlife Service indicated that the water quality standard would be met if flows in the Platte River were at or above 1,200 ft³/s. This benchmark flow is also used as part of the impact evaluation to compare alternatives.

To compare alternatives, the June, July, and August flow data for the Grand Island gage from the operations studies for each of the alternatives were entered into a spreadsheet. The alternatives were compared to the flow of 1,200 ft³/s on the basis of the daily flows for each month. The number of days out of the 48-year record in the operations studies that the flow exceeded 1,200 ft³/s in June, July, and August were totaled and used as the basis for comparing alternatives with the Present Condition. In addition, the probability of exceeding the temperature standard using the curve from Figure 148 was calculated, and those results were also used to compare the various alternatives to that of the Present Condition.

For the turbidity analysis, the TSS data from the sediment-vegetation model, better known as the SED-VEG model for short, were used as the basis for comparison. A series of regression relationships was developed to relate turbidity to TSS. The problem, as noted above was that there is a relatively poor relationship between turbidity and flow, and there is no overlap between the available TSS and turbidity data. In view of the former problem, a revised procedure has been developed for the turbidity analysis.

Figure 149 shows a time series plot of the flow and turbidity data for the Grand Island gage. In the preliminary impacts report, it was noted that the direct log-log regression of turbidity on flow only had an r^2 of 0.048, indicating that the regression explained only about 5 percent of the variation in turbidity. There was a much better relationship between turbidity and the color of the water. Color is usually a reflection of decaying organic matter in the water; predominant sources include humic acids, humates, and decaying aquatic plants (EPA, 1986), some of which may be suspended or colloidal and affect the turbidity of the water.

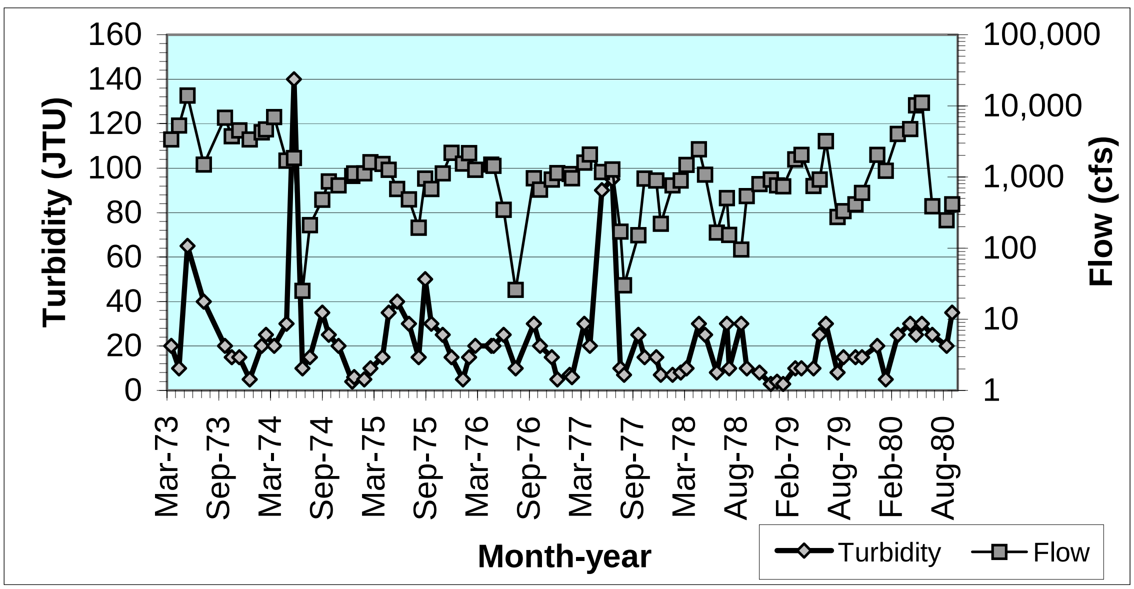


Figure 149. Time series plot of turbidity and flow at the Grand Island gage from 1973 through 1980

Figure 149 indicates that there is seasonality in the turbidity data. This is not surprising because of the strong influence on turbidity by color. Figure 149 also indicates that some of the turbidity peaks occur on the rising limb of the annual hydrograph, but that in many cases the flow continues to increase, while the turbidity decreases. In order to incorporate some of the other variation into the turbidity flow relationship, a multiple regression relationship was developed using, in addition to the natural logarithms of turbidity and flow, the month number and a season variable. The season as defined in the variable included 3 classes. The first class was the low-flow, usually ice-affected months of December and January. The third class included the runoff months of May and June. The remaining months were grouped into the intermediate group 2. The significance of the independent variables in the regression are shown in Table 54. Although each of the independent variables is statistically significant, the relationship with flow is the least significant of the 3 independent variables, based on the magnitude of the t-statistics for the coefficients. Alternatively, season is by far the most significant of the variables included in the regression. The only use for the multiple regression is to develop the best set of turbidity data possible for deriving a TSS–turbidity regression; so the fact that flow is only a minor factor is not important to the analysis.

Table 54. Statistics associated with the multiple regression for turbidity on flow

Effect	Coefficient	Std. Error	Std. Coef.	t	P(2 Tail)
Constant	-0.211871	0.468185	0	-0.453	0.65208
Month	0.059071	0.019206	0.263174	3.076	0.00285
Season	0.771565	0.106735	0.598079	7.229	< 0.00001
ln(Flow)	0.152030	0.053135	0.244444	2.861	0.00535

Table 55 summarizes the regression relationships for the data used in performing the alternatives comparison. The first equation is for the relationship between TSS and flow. This regression is not actually used, since TSS data are output from the SEDVEG model. The regression is based on the flow and TSS output from the SEDVEG model using all 61 years of daily data (22,282 observations).

Table 55. Summary of regression relationships between flow, TSS, and turbidity

Dependent variable	Independent variable	r^2	Equation
TSS	Flow	0.933	TSS = exp(a+b*ln(Flow)
Turbidity	Month, Season, Flow	0.445	Turbidity = exp(a+b*Month+c*Season+d*ln(Flow))
Turbidity	TSS	0.611	Turbidity=a*TSS$^{(b/TSS)}$

To perform the alternatives comparison based on the TSS output from the SEDVEG model, a relationship between TSS and turbidity had to be derived. The equation shown in Table 55 is based on a modified geometric fit. It is probably a more realistic approximation than an earlier derived regression (see Yahnke, 2001). The predicted values of turbidity from the multiple regression on season, month, and flow are plotted against TSS on Figure 150. The flows from the 1989-1999 TSS data set were used to develop the turbidity data. The measured TSS and calculated turbidity data were entered into the program “CurveExpert” to calculate the regression. The equation for the turbidity-TSS regression is also shown on Figure 150.

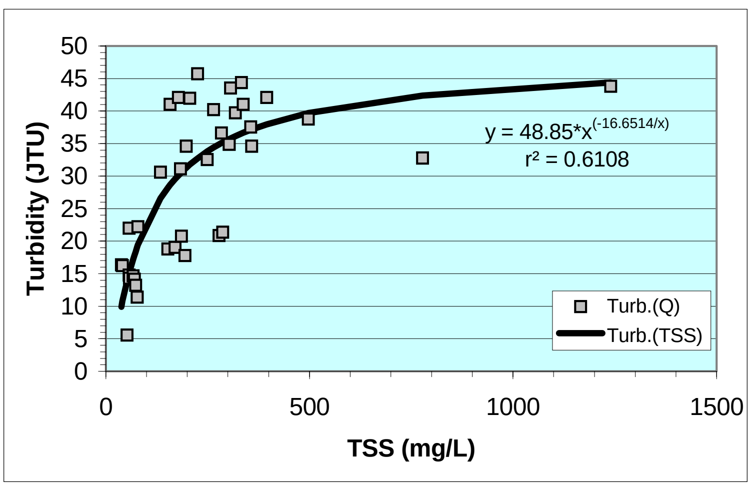


Figure 150. Relationship between turbidity and TSS used in the Central Platte impact analysis

The much lower r^2 for the relationship shown on Figure 150 than the regression derived in Yahnke (2001) is a result of the scatter in turbidity-TSS data used to develop the regression. Nevertheless, the equation generates a plot of the same general form as the log-log regression derived previously. The form of the relationship is a reflection of the fact that most of the effect on turbidity is due to fine sediments, as was described in the previous section of this report. At very high concentrations of TSS, most of the TSS is due to coarser particles that do not contribute to turbidity. In the instrument used to measure turbidity, these coarser particles would settle out before the measurement can be made.

Figure 150 indicates that the maximum turbidity that can be predicted from the regression is 45 JTU. Although Figure 149 indicates that there are measured turbidity values in excess of 45 JTU, these include only 4 of the measurements and could be considered statistical outliers. The vast majority of the measurements shown on Figure 149 are less than 45 JTU.

Present Condition

Temperature

As has been noted elsewhere in this appendix, the Present Condition as defined for the EIS has existed in the Middle Platte Basin since 1997, when the Biological Opinion for the Kingsley Dam hydro plant was issued and implemented. Recent temperature measurements made by the NDEQ in 2002 and 2003 from the Middle Platte at the Overton and Grand Island gages are shown on Figure 151. The warmwater temperature criterion of 90°F is exceeded in both years at Overton. There were no measurements made at Grand Island during 2002, but the criterion was exceeded during 2003 at that gage (Figure 151). These results that the warmwater temperature criterion is still exceeded during the summer in the Middle Platte Basin.

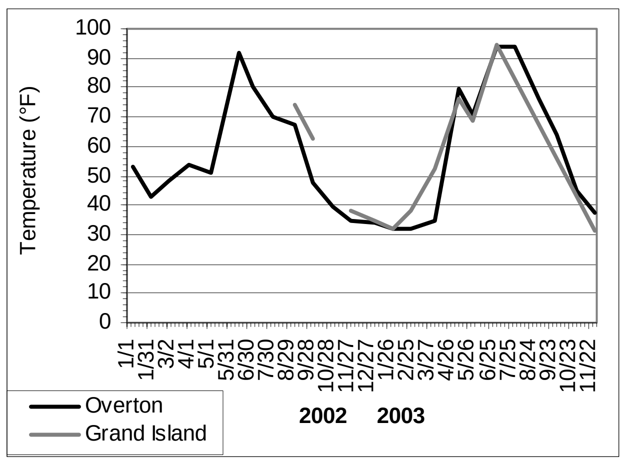


Figure 151. Platte River temperatures at Overton and Grand Island during 2002 and 2003

To create a basis for comparison of the various alternatives with the Present Condition, the various categories of temperature-related characteristics were calculated from the Platte River EIS hydrology model flow output for the months of June through August. The results are shown in Table 56. The flows are projected to exceed the 1,200 ft^3/s benchmark about for over ½ of the days in June, about ⅓ of the days in July, but less than 10 percent of the days in August. As is shown on Figure 151 by the measured data from 2003, there is still are reasonably high probability of exceeding the warmwater temperature criterion. The probabilities range from about 3 in 10 during June and July and over 4 in 10 during August (Table 56).

Table 56. Present Condition temperature-related baselines at Grand Island			
Category	June	July	August
Days with flow greater than 1,200 ft^3/s	744	519	114
Total number of days in the in the summary	1440	1488	1488
Percent of days with flow greater than 1,200 ft^3/s	51.7%	34.9%	7.7%
Probability of exceeding 32°C (90°F)	0.301	0.327	0.435

Turbidity

At the times that NDEQ measured temperatures at Overton and Grand Island, they also made turbidity measurements. These measurements are plotted on Figure 152. As was the case with the temperature measurements, there are two full years of turbidity measurements at Overton, but at Grand Island, there is only a partial year of data from 2002 and a full year from 2003. Although Overton will not be used in the alternatives comparison, the data from the site are included on Figure 152 to show that high turbidity readings at Overton do not necessarily translate into high readings at Grand Island. Although the turbidity readings are high at both Overton and Grand Island (80 and 120 NTU respectively) in the late summer of 2002, there are no coincidentally high turbidity measurements in 2003. The only turbidity readings greater than 40 JTU at the two sites during 2003 were made at Grand Island during the spring and at Overton in the very late summer (Figure 152). On the other hand, the turbidity data for the two sites nearly overlap during the winter of 2003.

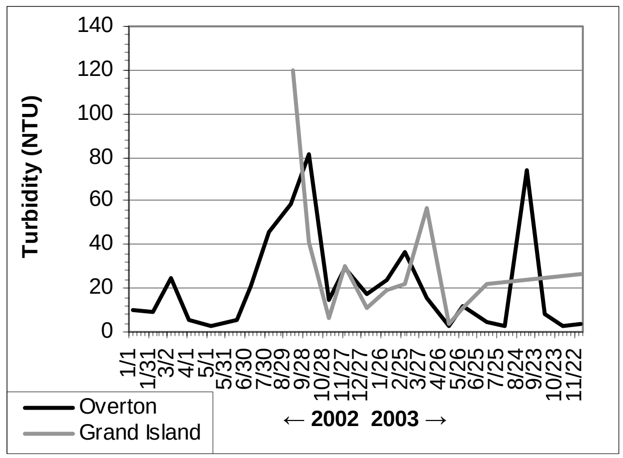


Figure 152. Platte River turbidity at Overton and Grand Island during 2002 and 2003

Figure 153 shows the cumulative frequency distribution of the historic turbidity data shown above on Figure 149. The cumulative frequency distribution of the Present Condition turbidity as calculated from the TSS estimates from the SEDVEG model is also shown on Figure 153. The Present Condition data used for the cumulative frequency distribution include only the water years 1973 through 1980, the period of record for the historic data. In

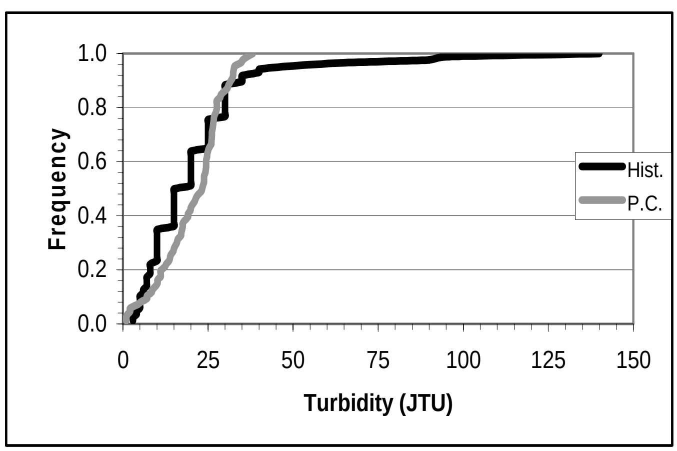


Figure 153. Cumulative frequency distributions of historic and Present Condition turbidity at Grand Island – 1973-1980

general, over most of the distributions, the Present Condition turbidity is somewhat higher than the historic data. The exception is at the upper end of the two distributions, where the observed data exceed the calculated Present Condition. The change occurs at about the 0.95 level on the frequency distributions and relates to the maximum estimated turbidity from the regression shown above on Figure 150. Within the distributions below 0.95, the median difference is 2 JTU, with a range of ±8 JTU. Considering that the historic data do not represent the Present Condition as defined in the Program, the differences in the distributions of turbidity data are not that great.

Effects of Alternatives

Temperature

Table 57 shows the percent of the days that the flow in the Platte River at Grand Island exceeds 1,200 ft^3/s for the Present Condition and each of the alternatives in the 48 years of the Platte River model simulations. The Governance Committee Alternative shows an increase over the Present Condition in the percent of the days of flow greater than 1,200 ft^3/s in both June and August and a slight decrease in the percentage in July. The same pattern by month as is shown by the Governance Committee Alternative is shown by each of the other alternatives, *i.e.* increases in the number of days of flow greater than 1,200 ft^3/s in June and August and a decrease in July. The changes in July and August are generally less than one percent. The overall percent of days with flow greater than 1,200 ft^3/s remains below 10 percent under any of the alternatives.

Table 57. Percent of days with flow greater than 1,200 ft^3/s at the Grand Island gage under the Present Condition and with each of the action alternatives

Month	Present Condition	Governance Committee	Full Water Leasing	Wet Meadow	Water Emphasis
June	51.7%	55.5%	62.9%	56.2%	60.8%
July	34.9%	33.3%	33.9%	34.6%	33.3%
August	7.7%	8.3%	8.1%	8.3%	8.6%

The differences from the Present Condition shown by the various alternatives differ among the three months (Table 57). The greatest increase in days with flow greater than 1,200 ft^3/s relative to the Present Condition in June would be with implementation of the Full Water Leasing Alternative, with the Water Emphasis Alternative also showing greater than 60 percent of the days of flow greater than 1,200 ft^3/s. The smallest decrease in days with flow greater than 1,200 ft^3/s relative to the Present Condition in July is projected to be with implementation of the Wet Meadow Emphasis Alternative, followed by the Full Water Leasing Alternative. Although none of the alternatives perform particularly well in providing flows in excess of 1,200 ft^3/s during August, the Water Emphasis Alternative performs the best, followed by the Governance Committee and Wet Meadow Emphasis alternatives. Based on the preceding, each of the alternatives can perform relatively well in providing flows of 1,200 ft^3/s, but the one that does the best at it varies with month.

Table 58 summarizes the results of the analysis of the probability of exceeding the warmwater aquatic life temperature criterion. In Table 58, the lower the probability, the better the result in terms of not exceeding the criterion. The complement of the probabilities in Table 58, *i.e.* 1 – probability, is the probability of meeting or not exceeding the temperature criterion.

Table 58. Probability of exceeding 32°C (90°F) at the Grand Island gage

Month	Present Condition	Governance Committee	Full Water Leasing	Wet Meadow	Water Emphasis
June	0.301	0.264	0.234	0.262	0.238
July	0.327	0.337	0.374	0.320	0.323
August	0.435	0.465	0.553	0.443	0.421

The results in Table 58 do not exactly show the same order of performance as was done with the 1,200 ft^3/s indicator in Table 57. Although all of the alternatives show a reduced probability of exceeding the temperature criterion in June, they do not show similar decreases in August, nor similar increases in July. For example, the Governance Committee, Full Water Leasing, and Wet Meadow Emphasis alternatives each show an increased probability of exceeding the standard relative to the Present Condition during August, while the Water Emphasis Alternative shows a reduced probability (Table 58). During July, the Governance Committee and the Full Water Leasing alternatives show increased probabilities of exceeding the criterion relative to the Present Condition, while the other two alternatives show a slightly reduced probability.

The results in tables 57 and 58 give very different measures of performance by the various alternatives relative to meeting the temperature criterion. For example, the results in Table 57 are based on meeting a benchmark flow. It is assumed that the criterion would be met at that flow. Alternatively, the probabilities a based on a continuum of flow. As was shown on Figure 148 above, the probability of exceeding the temperature criterion is never 1, no matter how low the flow gets, and never 0, no matter how high the flow may be. The lack of absolutes reflects the influence of other factors in determining whether or not the temperature criterion may be met. In other words, flow has an influence on water temperature, but it is by no means the only control on how warm the water gets. Nevertheless, there is both an overall increase in the frequency of exceeding 1,200 ft^3/s and a net decrease in the probability of exceeding the warmwater aquatic life temperature criterion with any of the alternatives. From that perspective, the Program would have a positive effect on water quality in the Middle Platte Basin.

Turbidity

Figure 154 shows plots of the median monthly turbidity of the Platte River in the vicinity of Grand Island for the Present Condition and each of the alternatives. The plots also show sets of error bars associated with each of the median monthly values. The error bars represent the 25th and 75th percentiles from the cumulative frequency distributions of the monthly data. Those percentiles approximate and the plotted error bars represent an approximation of plus or minus one standard deviation about the median (the 50th percentile).

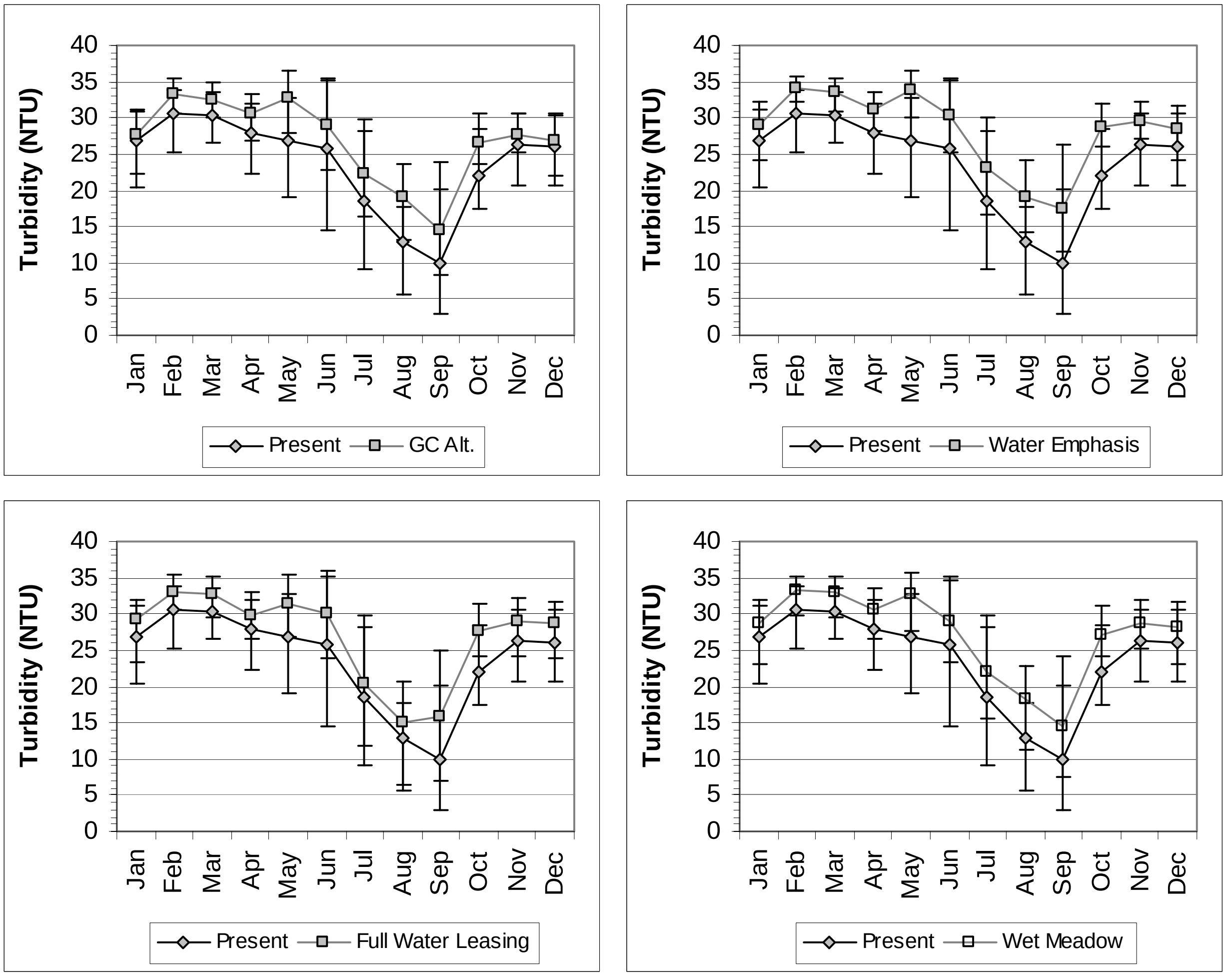


Figure 154. Mean monthly turbidity of the Present Condition and each of the action alternatives

Each of the alternatives leads to an increase in the median turbidity of all months of the year. Since one of the goals of the Program is sediment management, and a goal of sediment management is to increase the movement of sediments in the river, an increase in turbidity is an indicator that the Program is succeeding. The increase in turbidity would have to be considered a trade-off for the success of the Program.

Figure 154 also shows that the largest increases in turbidity occur in the late spring (May) and late summer (August and September) with all of the alternatives, although there is a substantial increase in turbidity throughout the summer with the Water Emphasis Alternative. However, the error bars for the Present Condition and each of the alternatives overlap during all months. The overlap would indicate that the increases in turbidity are well within the normal range of turbidity in all months. The increases in turbidity would be a shift from what would be about the center of the current monthly distributions of turbidity to what is now represented by some higher level within that distribution. Despite the shift, the increases in turbidity would not be significant from a statistical perspective. In other words, the increases in turbidity would probably not be noticeable to most people.

It is difficult to evaluate the potential effects of increased turbidity on aquatic life. As an example, an analysis of the hypothetical effects of increased turbidity on small fish has been performed. Miner and Stein (1996) evaluated the effects of turbidity on the reaction distance of small bluegills to a variety of predators. The bluegills used in the study were 41.2 ± 0.3 mm or about 1½ inches. The general effect of turbidity on reaction distance was defined by the regression equation: $y = 98.2x^{-0.624}$, where y is the reaction distance and x is the turbidity of the water in NTU. The results of the analysis based on the daily turbidity data for May from the above summarized results are shown in Table 59. The median change in reaction distance indicates that the reaction distances with each of the alternatives would decrease by about an inch (2.5 cm = 1 inch), but slightly more with the Water Emphasis Alternative and as much as 76 cm (30 inches). These large changes would be rare, and most changes (> 95 percent based on daily data) would be much smaller. The problem with the above is that it may or may not be applicable to Platte River forage fish. Although fish used in the experiment were of about the correct size, bluegills are not minnows like the forage fish of most concern in the Middle Platte Basin. How applicable such an effect might be to Platte River forage fish is unknown.

Table 59. Changes in reaction distance (cm) due to increased turbidity

	Governance Committee	Water Emphasis	Full Water Leasing	Wet Meadow
Maximum	-55	-76	-55	-68
Median	-3	-4	-3	-3
Minimum	-1	-2	-1	-1

Ground Water Quality

A large mound of ground water has developed near the Platte River in south-central Nebraska near the study area (Figure 155). The mound is characterized by a rise of more than 50 feet in the ground water surface elevation since a pre-development bench mark. The mound underlies most of Gosper, Phelps, and Kearney counties (Figure 156). The mound lies just south the Platte River and just north of the Republican River. Because of its proximity to the Platte River, it is a potential source of water for augmentation flows. This section evaluates the quality of the ground water in the mound.

The Nebraska Natural Resource Commission has a website that has a data base of water data for the State. Figures 155 and 156 were downloaded from the website. The outline of the mound was overlaid onto a map showing the location of wells with ground water quality data, which was also downloaded from the site (Figure 157). Data summaries for the wells within the mound were downloaded from the website. A summary of all of the water quality data within the mound is shown in Table 45.

The water quality of the mound appears to be generally good (Table 45). The TDS and specific conductance for the mound is somewhat lower than that of the Platte River. The water is predominantly a calcium-bicarbonate-sulfate type based on the average composition shown in Table 45.

As was noted above, the retrieval of ground water data included only summaries - means, minima, and maxima. The summary data in Table 45 include the total number of samples within the summaries, the overall maximum value for each parameter, the average of the maximum values for each of the wells, the overall minimum for all wells, and a mean calculated by multiplying the average for each well by the total number of samples from the well and dividing that sum by the total number of observations for all wells. For most parameters the mean maximum and the overall mean are similar; this is because the majority of wells have only been sampled once.

Table 45 indicates that the most noteworthy water quality problem in the ground water in the mound is excessive nitrate. It should be noted that the first line of nitrate data in Table 1 is expressed as nitrate while the remaining 3 expressions of nitrate are as nitrogen. If the data expressed as nitrate are converted to an equivalent concentration of nitrogen as shown in the footnote to Table 45, the results are similar to the other forms; *i.e.* the maximum is 47, the mean maximum is 5.6, and the mean is 4.6 mg/L. The samples expressed as nitrate were collected between 1936 and 1978; the samples summarized on the next line of Table 1 were collected in 1969-70; the nitrite plus nitrate samples were all collected between 1975 and 1989. Based on the results above and in Table 45, there has been little change in nitrate concentrations in the ground water mound on the average. However, the maximum of the nitrite plus nitrate samples in the more recent data is less than ½ that in the earlier nitrate data.

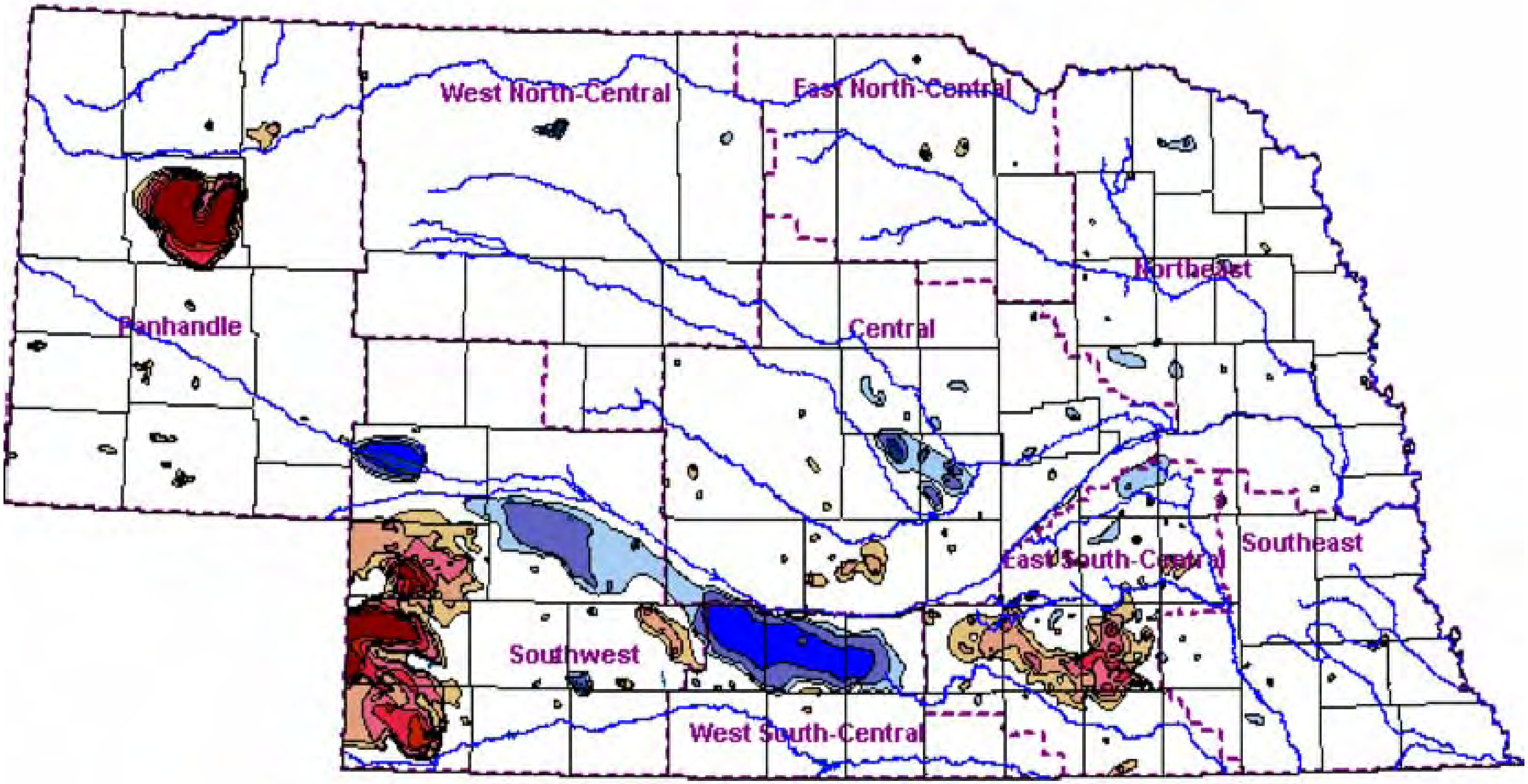


Figure 155. Location of the ground water mound (dark blue area) under south central Nebraska in the Big Bend area of the Platte River (adapted from Nebraska NRC data bank)

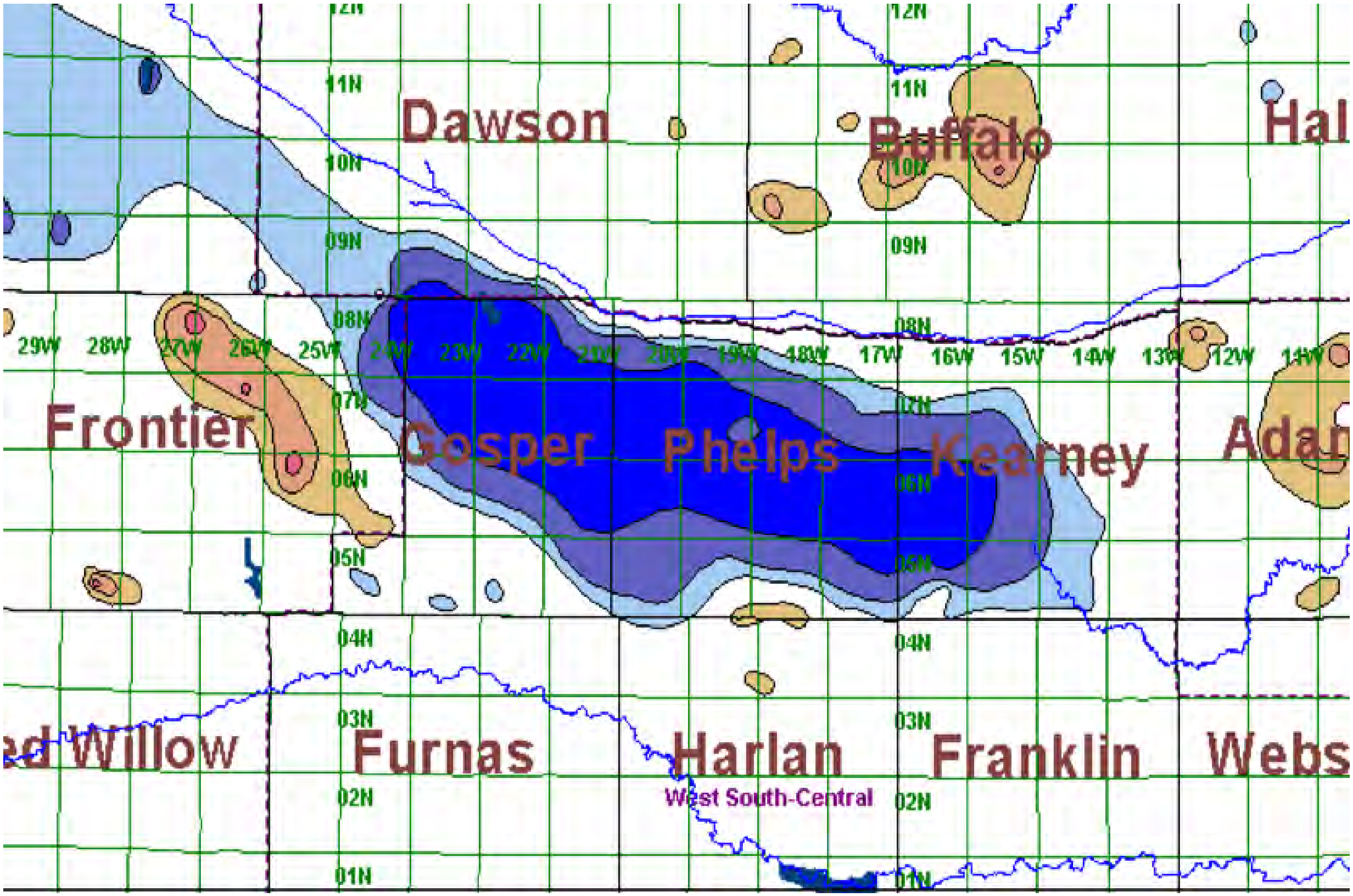


Figure 156. View of location of ground water mound showing county boundaries and townships and ranges

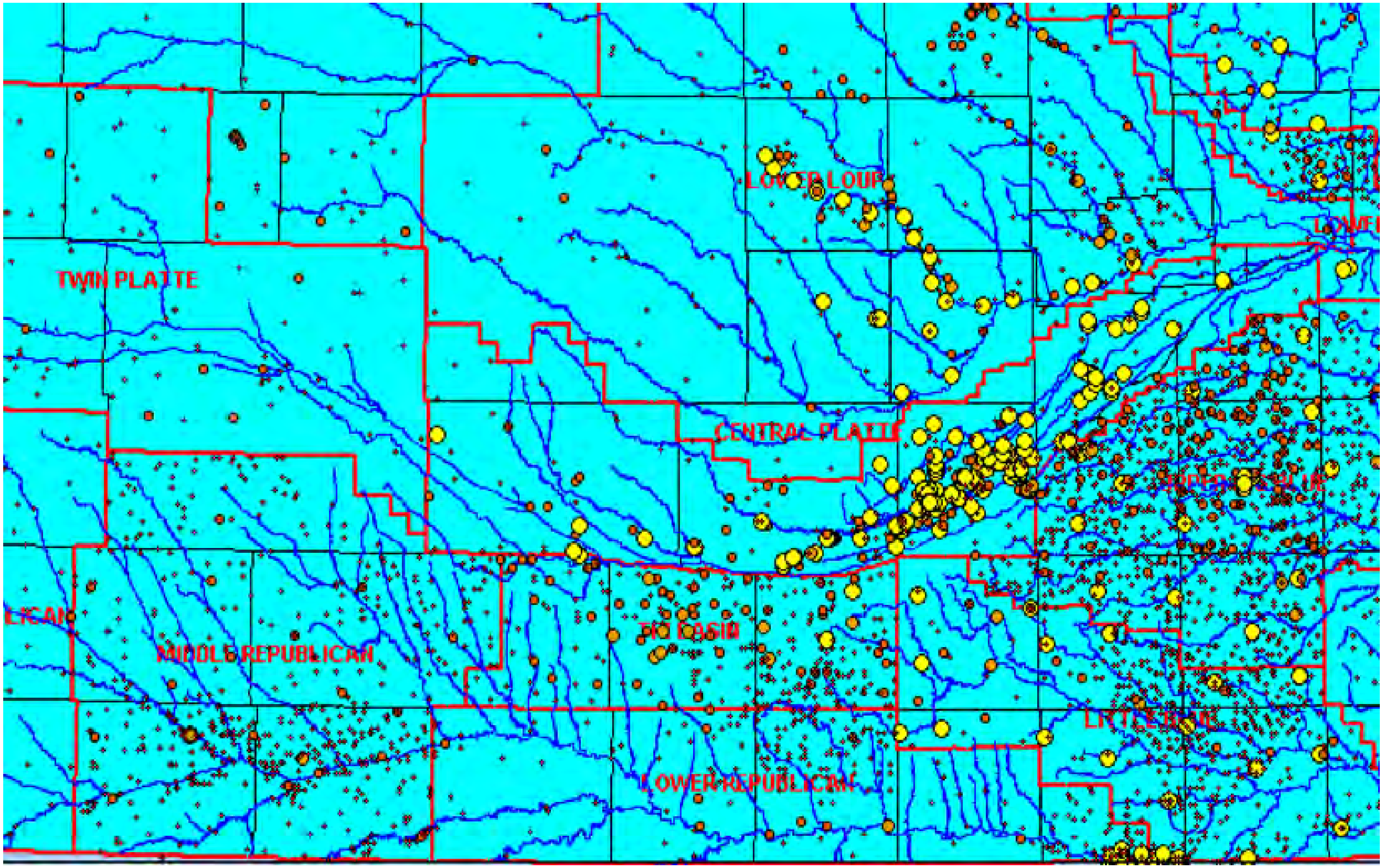


Figure 157. Location of wells in south central Nebraska with water quality data; natural resource district boundaries are also shown

Table 45. Summary of Water Quality Data from the Ground Water Mound since 1949

	No. of Obs.	Maximum	Mean Max.	Minimum	Mean
Temperature, Water (°C)	84	24	14.4	10.6	14.3
Specific Conductance (µmho/cm @ 25c)	127	1460	838	353	798
pH (Standard Units)	104	8.6	7.5	6.5	7.4
Bicarbonate Ion (mg/L as HCO_3)	58	377	275	48	269
Calcium, Dissolved (mg/L as Ca)	74	210	115	13	107
Chloride, total in Water (mg/L)	75	89	24	2	22
Hardness, Total (mg/L as $CaCO_3$)	59	583	314	78	303
Magnesium, Dissolved (mg/L as Mg)	73	24.7	15.9	5.7	15.2
Nitrate Nitrogen, Dissolved (mg/L as NO_3[1])	47	208	24.9	0	20.4
Nitrate Nitrogen, Total (Mg/L as N)	32	47	6.3	0	5.0
Nitrite plus Nitrate, Diss. (mg/L as N)	36	18	4.6	0	4.1
Nitrite plus Nitrate, Total (mg/L as N)	30	20	7.6	0.01	7.1
Phosphorus, Dissolved (mg/L as P)	21	0.35	0.2	0	0.2
Potassium, Dissolved (mg/L as K)	70	38	13.4	4	13.1
Residue, total Filtrable (Dried at 180°C, mg/L)	44	1150	497	237	491
Sodium, Dissolved (mg/L as Na)	70	71	35	9.1	34
Solids, Dissolved-sum of Constituents (mg/L)	58	1030	487	223	480
Sulfate, Total (mg/L as SO_4)	96	364	148	3	138
Aluminum, Dissolved (µg/L as Al)	15	200	131	30	122
Arsenic, Dissolved (µg/L as As)	6	10	2.8	0	2.0
Boron, Dissolved (µg/L as B)	60	300	110	0	89
Cadmium, Dissolved (µg/L as Cd)	23	13	0.9	0	0.8
Cobalt, Dissolved (µg/L as Co)	23	2	0.1	0	0.2
Copper, Dissolved (µg/L as Cu)	13	30	4.4	0	3.8
Iron, Dissolved (µg/L as Fe)	48	390	66	0	51
Lead, Dissolved (µg/L as Pb)	31	46	2.4	0	1.5
Lithium, Dissolved (µg/L as Li)	13	40	22	0	24

Table 45. (Continued)

	No. of Obs.	Maximum	Mean Max.	Minimum	Mean
Manganese, Dissolved (µg/L as Mn)	23	380	30	0	39
Nickel, Dissolved (µg/L as Ni)	30	16	2.0	0	2.0
Selenium, Dissolved (µg/L as Se)	29	103	20	1	13
Silver, Dissolved (µg/L as Ag)	25	0	0.0	0	0.0
Strontium, Dissolved (µg/L as Sr)	17	770	541	220	512
Zinc, Dissolved (µg/L as Zn)	13	30	12	0	11
2,4-D in Whole Water Sample (µg/L)	2	0.5	0.5	0.5	0.5
Alachlor Whole Water, (µg/L)	8	0.3	0.2	0.1	0.2
Ametryne (Gesapax or Evik) Total (µg/L)	6	0.1	0.1	0.1	0.1
Atrazine (Aatrex) in Whole Water Sample (µg/L)	17	0.8	0.3	0.04	0.3
Cyanazine in the Whole Water Sample (µg/L)	16	0.4	0.4	0.1	0.3
Metolachlor, Whole Water, Total Recoverable (µg/L)	3	0.1	0.1	0.1	0.1
Metribuzin, Whole Water, Total Recoverable (µg/L)	3	0.1	0.1	0.1	0.1
Picloram in Whole Water Sample (µg/L)	12	0.8	0.1	0	0.2
Prometone in Whole Water (µg/L)	16	0	0.0	0	0.0
Prometryne in Whole Water (µg/L)	6	0	0.0	0	0.0
Propazine, Coulson Conductivity, water (µg/L)	16	0.11	0.1	0.04	0.1
Simazine in Whole Water (µg/L)	16	0	0.0	0	0.0
Simetryne in Whole Water (µg/L)	6	0	0.0	0	0.0
Treflan, Microcoulometric, water (µg/L)	3	0.1	0.1	0.1	0.1
Depth to Water Level (Feet below Land Surface)	6	80	48	12	48
Depth, Total of Well (Ft below Land Surface Datum)	72	338	178	66	170
Elevation of Land Surface Datum (Ft. above MSL)	34	2550	2330	2155	2330

[1] To convert NO_3^{-1} to N multiply by 0.226 (the atomic weight of nitrogen [14] divided by the ionic weight of NO_3^{-1} [62])

Maximum concentrations of several trace elements also appear somewhat elevated in the ground water mound. However, all of the trace element data, with the exception of iron and manganese, are from 1969 and 1970. There are several trace elements that exceed aquatic life standards. These are not usually a concern for ground water, but if the program provides water from the mound for irrigation in exchange for Platte River water, the potential effect of return flows on fish and wildlife propagation are considered, then the aquatic life criteria become a concern. Based on Table 45, trace elements that may exceed aquatic life criteria include cadmium, copper, and selenium.

As a follow-up to the screening based on data summaries, the actual data for the wells that made up the summaries were retrieved. These were compared to various water quality standards, primarily drinking water MCL's and SMCL's (MCL = maximum contaminant levels and secondary MCL's). The comparison is summarized in Table 46. The summary table indicates that more than ½ of the TDS (residue) samples exceed the SMCL for drinking water. A small percentage of the samples exceed the MCL's for nitrate (NO_3-N) and cadmium. None of the samples exceeded the standards for copper or lead. A large percentage of the samples exceeded the SMCL for aluminum and the aquatic life criterion for selenium. These latter 2 elements will be evaluated further.

Table 46. Comparison to water quality standards based on pre-1980 data.

	Residue Diss-180°C (mg/L)	NO_3-N Total (mg/L)	Cadmium Cd, diss (µg/L)	Copper Cu, diss (µg/L)
Minimum	237	0	0	0
Median	528	2.9	0	1
Maximum	1150	47	13	30
No. of Obs.	45	25	21	10
No. > Std.	26	2	1	0
Standard	500	10	5	67
Use	SMCL	MCL	MCL	Aq.L.
	Lead Pb, diss (µg/L)	**Aluminum Al, diss (µg/L)**	**Lithium Li, diss (µg/L)**	**Selenium Se, diss (µg/L)**
Minimum	0	30	18	< 1
Median	0	100	30	10
Maximum	46	200	40	103
No. of Obs.	21	14	10	29
No. > Std.	0	12	NA	19
Standard	50	50	NA	5
Use	MCL	SMCL	NA	Aq.L.

The most common detection limit for aluminum, despite the minimum concentration of 30 µg/L, was 100 µg/L. Obviously, the usual detection limit is greater than the SMCL. The problem with detection limits is that the greatest error in a chemical analysis is at or near the detection limit. In this data set, 8 of the 14 samples that exceeded the SMCL for aluminum were equal to 100 µg/L. This raises the probability that the number of samples that actually exceeded the standard is less than the 12 shown in Table 46.

Selenium has been a serious concern to the Department of the Interior since the discovery of large scale waterfowl mortality, birth defects, and avian mortality at Kesterson Reservoir, part of the Kesterson National Wildlife Refuge, on the San Luis Drain in Central California in 1982 (Deason, 1986). As is shown in Table 46, the median selenium concentration in the ground water in the mound is twice the EPA chronic aquatic life criterion. However, all but one of the samples collected in the ground water mound and summarized in Table 46 were collected between 1969 and 1972. In 1981, the USGS, the source of all of the ground water quality data, issued Quality of Water Technical Memorandum No. 81.12, which advised "that values of dissolved and total selenium exceeding 5 µg/L in samples collected before 1975 should be used great caution." The only selenium sample that was collected after 1975 (collected in 1978) was the one showing the minimum in Table 46.

Because of concerns related to possible high selenium in the ground water mound and the uncertainty surrounding the validity of the data, additional samples for water quality analysis were collected from 29 wells between the 10^{th} and 19^{th} of January 2000. Twenty-eight of the samples were analyzed for trace elements (Michael E. Slifer, District Chief, USGS, Lincoln, Nebraska, Letter of March 22, 2000, to Jim Yahnke, Bureau of Reclamation, Denver, Colorado). The types of analyses and a summary of the results are shown in Table 47. The table also shows a comparison of the data to State of Nebraska water quality criteria or standards. Since most of the concern in this investigation relates to wildlife, the aquatic life criteria are shown in the table where they exist. In the absence of aquatic life criteria, other standards or criteria are used. The reason for the preferential use of the aquatic life criteria relates to the potential use of the ground water as an agricultural supply in exchange for providing additional flows in the Platte River. If the ground water exceeds the aquatic life criteria, return flows from the agricultural use of the ground water would also likely exceed the aquatic life criteria, with potential adverse effects. Even if the irrigation water did not exceed any criteria, the return flows could still do so, because the irrigation water would be further concentrated through evapotranspiration.

Five of the standards and criteria used on Table 47 are based on the protection of aquatic life use of the river. The temperature standard is not even approached, nor are the criteria for copper or nickel. However, the selenium criterion is exceeded in 12 of the 28 samples. The selenium results provide a degree of verification of the previous data set for selenium; in the 1969-78 data set, 66 percent of the samples exceeded the selenium criterion, while in the January 2000 data set, about 43 percent of the samples, or nearly ½ the well samples, exceeded the selenium criterion.

Although the focus of the standards and criteria used in the data evaluation was aquatic life, 11 of those in Table 47 are drinking water standards. Since many of the wells are municipal or domestic wells (14 of the total), the drinking water standards are more appropriate for many of them.

There were no exceedences of the standards for arsenic, barium, cadmium, chromium, lead (usually more of a problem in distribution systems of older homes), silver or zinc. There were 2 samples above the upper limit of the pH range for drinking water. Nitrate has been a noted problem in the ground water in the Central Platte for quite some time (Engberg and Spalding, 1978). Six of the 29 (about 20 percent) NO_2+NO_3 samples exceeded the drinking water standard of 10 mg/L. About 10 percent of the manganese samples exceeded the drinking water standard; excessive manganese may impart a metallic taste to water and may also cause staining of clothes when exposed to detergents. About 20 percent of the uranium samples also exceeded the uranium drinking water standard.

Table 47. Summary of January 2000 chemical analysis of samples from the ground water mound

	Temperature [°C]	Sp. Cond. [μS/cm]	D.O. (mg/L)	pH	NO_2+NO_3 (mg/L)	Arsenic (μg/L)	Barium (μg/L)
Minimum	10.5	152	0.2	6.9	< 0.05	0	11
Median	13.5	820	0.8	7.2	2.76	4.5	143
Maximum	18.9	1660	8.1	10.3	20.4	45	495
No. of Obs.	29	29	28	29	29	28	28
No. > Std.	0	0		2	6	0	0
Standard	32	2000	N/A	6.5-8.5	10	50	1000
Use Class.	Aq.L.	Ag.		D.W.	D.W.	D.W.	D.W.

	Beryllium (μg/L)	Boron (μg/L)	Cadmium (μg/L)	Chromium (μg/L)	Cobalt (μg/L)	Copper (μg/L)	Lead (μg/L)
Minimum	<1.0	47	<1.0	< 0.80	<1.0	1.5	<1.0
Median	<1.0	64	<1.0	0.72	<1.0	5.0	<1.0
Maximum	<1.0	179	<1.0	5.00	3.1	15	43
No. of Obs.	28	28	28	28	28	28	28
No. > Std.			0	0		0	0
Standard	N/A	N/A	5	100	N/A	66.9	50
Use Class.			D.W.	D.W.		Aq.L.	D.W.

	Manganese (μg/L)	Thallium (μg/L)	Molybdenum (μg/L)	Nickel (μg/L)	Silver (μg/L)	Strontium (μg/L)	Vanadium (μg/L)
Minimum	<1.0	< 0.9	1.5	<1.0	<1.0	36	<1
Median	1.7	< 0.9	3.6	3.4	<1.0	586	12
Maximum	1730	< 0.9	59	13	<1.0	943	40
No. of Obs.	28	28	28	28	28	28	28
No. > Std.	5			0	0		
Standard	50	N/A	N/A	18.6	50	N/A	N/A
Use Class.	D.W.			Aq.L.	D.W.		

	Zinc (μg/L)	Antimony (μg/L)	Aluminum (μg/L)	Lithium (μg/L)	Selenium (μg/L)	Uranium (μg/L)	Triazines (μg/L)
Minimum	6.3	<1.0	<1.0	11	<1.0	<1.0	<0.05
Median	45	<1.0	<1.0	25	3.5	7.1	0.08
Maximum	1600	<1.0	34	70	31	177	0.84
No. of Obs.	28	28	28	28	28	28	29
No. > Std.	0		0		12	4	0
Standard	2544	N/A	50	N/A	5	20	3
Use Class.	Aq.L.		D.W.		Aq.L.	D.W.	D.W.

NOTE - μS/cm is the metric equivalent to the traditional μmho/cm. The conversion factor is 1.
Use classes are as follows: Ag. – Agriculture
Aq.L. – Aquatic Life
D.W. – Drinking Water

Table 48 shows the results of a statistical comparison of the January 2000 samples with the data from the earlier samples retrieved from the NRC database. The comparison is based on a Mann-Whitney test performed in the statistical analysis package SYSTAT, version 9 (SPSS, 1999). The Mann- Whitney test, as performed in SYSTAT produces a X^2 statistic, each of which is shown in Table 48 for the analytes and physical measurements common to the two data sets. Based on the results in Table 48, only aluminum shows a difference between the two sampling periods. Recall that the detection limit for aluminum in the earlier data set was 100 µg/L. The detection limit in the January 2000 data set was 1 µg/L; the median concentration was below this detection limit, *i.e.* < 1µg/L. The maximum aluminum concentration in the January 2000 samples was 34 µg/L, which is near the minimum of the earlier data. These differences do not necessarily mean that the aluminum in the ground water has declined, but rather the analytical technique for the determination of aluminum has improved.

Table 48. Comparison of Nebraska NRC database to Jan. 2000 data for ground water mound

Variable	**X^2 Stat.**	**Prob. > X^2**
EC	1.10	0.293964
Aluminum	22.27	0.000002
Copper	3.68	0.054923
Lead	0.26	0.612602
Lithium	0.25	0.619015
NO_3-N	0.60	0.438223
Selenium	2.84	0.091739
Well depth	1.77	0.183170

As was noted earlier, the main reason for recent water quality sampling of the ground water mound was because of the high selenium in the earlier samples. The comparison in Table 48 indicates that there is no significant difference in the two data sets (probability > X^2 > 0.05), despite the apparently large differences between the respective median and maximum selenium concentrations. The Mann-Whitney test is performed on the ranks of the individual data points and the magnitude of the values is not a factor in the result. What this means is that there is enough overlap in the two sets of samples that they could easily be from the population of data. On this basis the high concentrations of selenium have been present in the ground water in the area for over 30 years.

Druliner and McGrath (1996) developed a regression relationship for predicting NO_3–N contamination in ground water in the Great Bend area to the north of the Platte River. The most significant independent variable in the NO_3–N regression was specific conductance. Other independent variables in the regression included the average hydraulic conductivity of the unsaturated zone and the median well completion data for wells within a 1-mile radius. The only one of these variables for which there are data in the January 2000 data set is specific conductance. The correlations between specific conductance and NO_3–N in the January 2000 data set, along with similar correlations in the earlier well data and a combined data set, are shown in Table 49. All of the correlations are statistically highly significant. In their study Druliner and McGrath (1996) concluded that specific conductance was a surrogate measure of NO_3–N, which was a major anion in the study area. However, NO_3–N is not a major anion in the ground water mound (see table 45); nitrate is slightly lower than chloride, which is well below concentrations of both bicarbonate and sulfate on the average. In the ground water mound it seems more likely that the higher specific conductance in the ground water is a reflection of the effects of recharge from the deep percolation component of irrigation water, which concentrates

Table 49. Correlations among EC, NO_3-N, and Selenium based on three different data sets					
Variable 1	Variable 2		Early Data	Jan 2000	Combined
Sp. Cond.	NO_3-N	r	0.741664	0.572610	0.592637
		Prob. > r	0.000022	0.001451	0.000003
		n	25	28	53
Selenium	NO_3-N	r	0.932711	0.686307	0.812710
		Prob. > r	< 0.000001	0.000077	< 0.000001
		n	15	27	42
Selenium	Sp. Cond.	r	0.653361	0.469431	0.467221
		Prob. > r	0.000122	0.013498	0.000283
		n	29	27	56

the recharge to a greater extent the more irrigation there is in an area. The NO_3–N is then a reflection of the application of fertilizer that is obviously also associated with irrigated agriculture. The association with irrigated agriculture in the Druliner and McGrath (1996) model was shown by the surrogate variable related to the well completion date. The conclusion was that sites that had been under irrigation longest had an increased probability of ground water contamination by NO_3–N.

Table 49 also shows correlations of selenium with both specific conductance and NO_3–N. The correlations of NO_3–N with specific conductance are better than those of selenium with specific conductance. However, the best correlations of any of those shown in Table Q5 are between selenium and NO_3–N. Walker (1999) has showed an increase yield of selenium from marine shales from a site near Grand Junction, Colorado, with increased addition of fertilizer-simulated nitrate. Given the magnitude of the correlations, this seems a likely possibility in this area of Nebraska as well.

The soils in the study area are primarily of loessial origin (Bowman, 1973; Wahl, 1981; Wahl *et al.*, 1984). The loess is underlain by the Ogallala Formation, which is primarily composed of calcareous silt, silty or sandy clay, and fine– to medium–grained sand (Peckinpaugh *et al.*, 1987). Bedrock consists of Cretaceous beds of shale with some thinner beds of shaley chalk and chalk (*ibid.*). Bedrock units are usually Pierre Shale or the Niobrara Formation where the Pierre Shale has been eroded away. Either of these formations may be seleniferous in Nebraska (Rosenfield and Beath, 1964). For the source of the selenium to be the bedrock, the selenium would be expected to be picked up as the water flowed across it. This would involve extensive lateral flow. The ground water mound has been formed in place in the unconfined aquifer and lateral flow should be limited. On this basis the selenium in the ground water should be derived directly from recharge into the mound. Since the Ogallala Formation is not known to be seleniferous [based on the inventory in Rosenfield and Beath (1964)], the source of selenium would be from the soil. Loessial soils may be seleniferous if the parent material were seleniferous. The parent material for the soil is shown as the Peoria Loess in Bowman (1973), Wahl (1981), and Wahl *et al.* (1984). Since this is a wind-deposited material, its actual source is unknown.

The selenium data from the January 2000 samples were mapped. These are shown on Figure 158. The data indicate that the area with excessive selenium seems to be generally confined to the center of the study area. The fringes to the east and west have low (below the EPA water quality criterion) to below detectable concentrations of selenium. Because of this, it may be possible to select areas that would be suitable for the recharge option to be implemented. Areas that would be excluded due to elevated ground water selenium would include essentially all of Phelps County, the eastern fringe of Gosper County, and the western ½ of Kearney County. Outside of these areas, the selenium concentrations are below toxicity effect levels.

Expanded Database for the Tri-County Area

The earlier selenium data were also mapped. It was noted on the map that the areal distribution of the earlier selenium data were confined to Phelps County and range 16W in Kearney County. The earlier STORET retrieval was made by entering the well numbers for wells in the Nebraska NRC data base. Another STORET retrieval was made based on the three counties that essentially define the areal extent of the ground water mound. This retrieval expanded the number of wells considerably. The retrieval included all of the parameters that were included in the January 2000 samples. The 2nd retrieval included many more water quality parameters than the initial one and expanded the number of observations, particularly for nitrate and selenium. The complete pre-1990 selenium set data is mapped on Figure 159, and the selenium data retrieved from STORET in the second attempt are summarized in Table 50.

The data in Table 50 also include a comparison to the same water quality standards as was done in Table 47 for the January 2000 data. The results show that the water is suitable for agricultural uses based on EC and boron. It should be noted that there is no Nebraska water quality standard for boron. The comparisons in tables 47 and 50 show that boron is not a problem in the area. The maximum boron concentration in either table is less than ½ the irrigation guideline from Ayers and Westcot (1985).

Table 46 showed exceedences of various water quality standards by nitrate, aluminum, selenium, and zinc. Table 47 showed no exceedences for aluminum, but did show some for nitrate and selenium. In addition to nitrate and selenium, manganese, uranium and zinc exceeded their respective water quality standards in some samples (all in the case of zinc).

In Table 47 pH was outside the range of the standard in 2 samples. Table Q6 shows both the minimum and the maximum to be outside the range of pH given by the water quality standard. Both of the values outside the pH range in Table 47 were above 8.5. The maximum in Table 50 is just barely outside the range. The other 2 values outside the ranges were the minimum and one other value that was just below the lower limit of the standard, *i.e.* 6.47. Both of the low values were observed on August 1987.

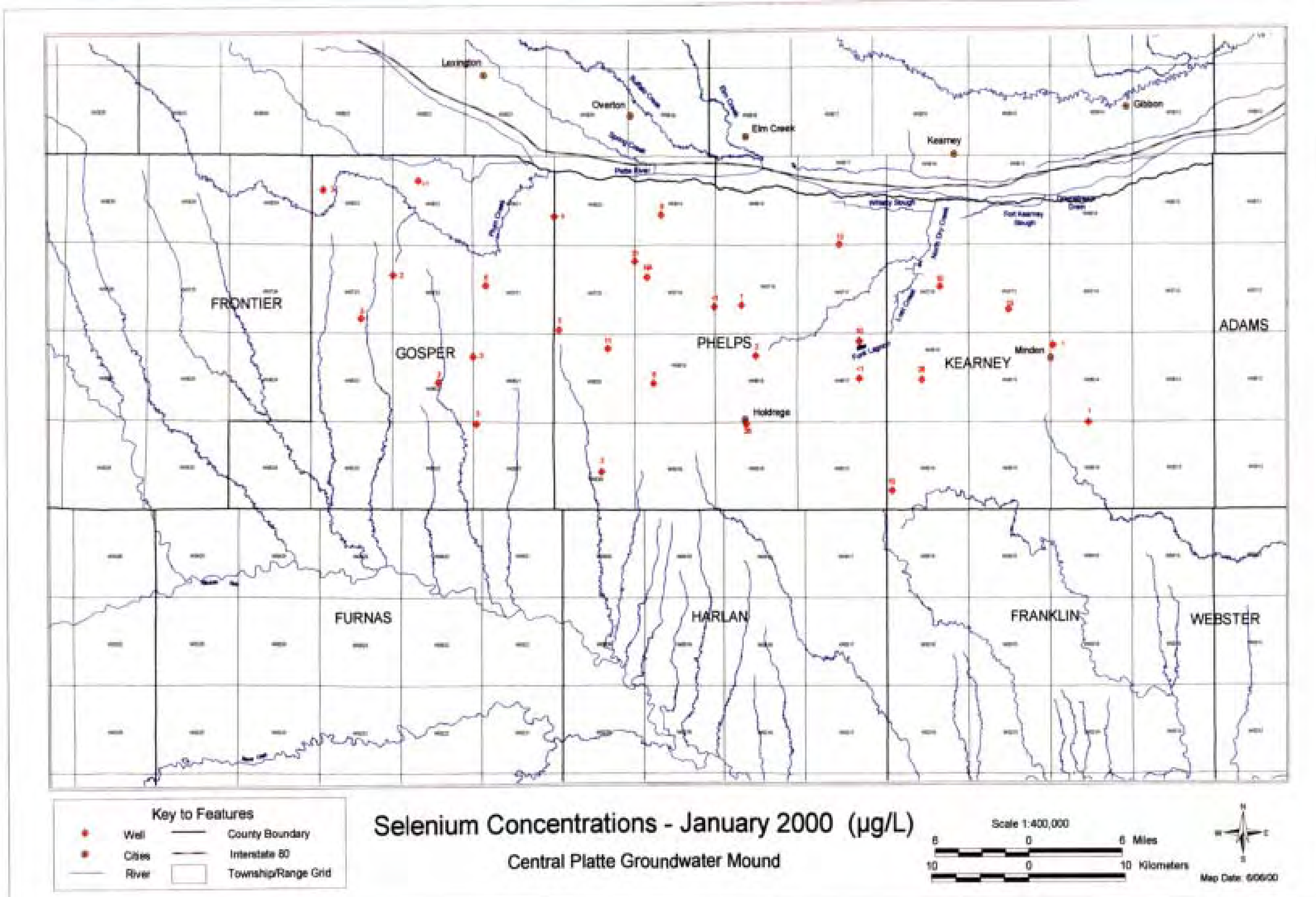


Figure 158. Location of wells and selenium concentrations in the 2000 Ground Water Mound data set

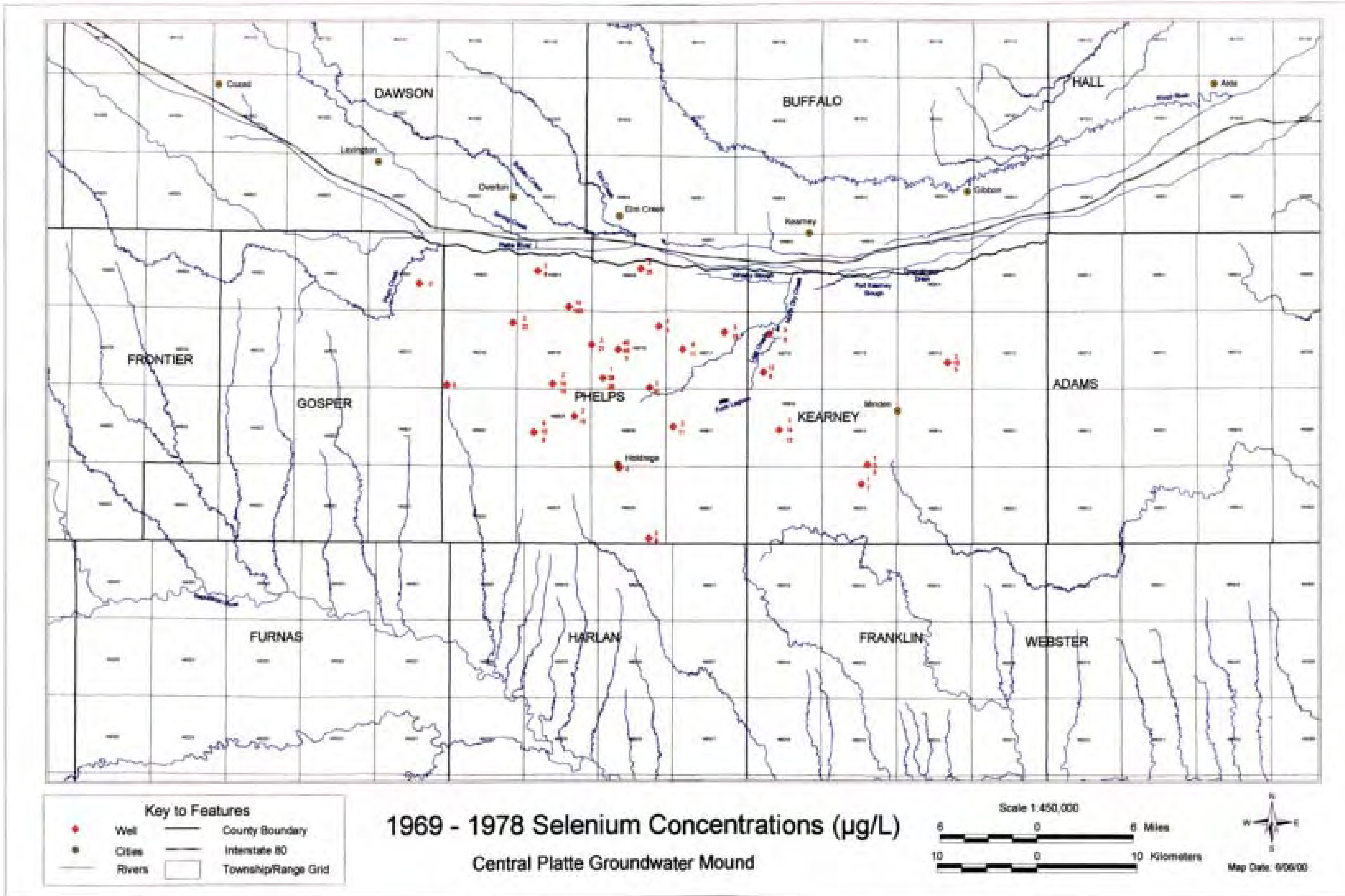


Figure 159. Location of wells and selenium concentrations in the 1969-1978 Nebraska Natural Resources Commission data set

Table 50. Summary of Ground Water Quality Data for Gosper, Kearney, and Phelps Counties from 1947-1989

	Sp. Cond. [µmho/cm]	**DO [mg/L]**	**pH**	**NO_3–N [mg/L]**	**Aluminum [µg/L]**	**Arsenic [µg/L]**
Minimum	170	2.4	5.11	0	0	0
Median	657	4.3	7.45	4	100	0
Maximum	1970	6.1	8.6	50	200	10
No. of Obs.	258	2	218	147	26	12
Standard	2000	NA	6.5-8.5	10	50	50
No. > Std.	0		3	12	22	0
Use	Ag.		MCL	MCL	SMCL	MCL
	Barium [µg/L]	**Boron [µg/L]**	**Chromium [µg/L]**	**Cobalt [µg/L]**	**Copper [µg/L]**	**Lead [µg/L]**
Minimum	0	0	0	0	0	0
Median	150	80	20	0	2	0
Maximum	300	300	20	2	30	46
No. of Obs.	6	131	3	44	25	44
Standard	1000	700	100	NA	66.9	50
No. > Std.	0	0	0		0	0
Use	MCL	Irr.	MCL			MCL
	Lithium [µg/L]	**Nickel [µg/L]**	**Selenium [µg/L]**	**Silver [µg/L]**	**Vanadium [µg/L]**	**Zinc [µg/L]**
Minimum	0	0	0	0	0	0
Median	30	0	7	0	4	10
Maximum	50	3	103	0	13	80
No. of Obs.	25	44	54	44	17	23
Standard	NA	18.6	5	50	NA	2544
No. > Std.		0	29	0		0
Use		Aq.L.	Aq.L.	MCL		Aq.L.

Use classes:
- Ag. Agriculture
- Irr. Irrigation guideline for sensitive crops (not a std.)
- Aq.L. Aquatic Life - Hardness = 300 mg/L $CaCO_3$
- MCL Primary Drinking Water Standard
- SMCL Secondary Drinking Water Standard

The detection limits for aluminum in most of the earlier samples was 100 µg/L. In older data no matter what the detection limit might be, low concentrations of trace elements tend to be clustered near it. In such data sets, these concentrations may seem high by today's analytical standards. In the case of aluminum the current water quality standard is ½ the earlier detection limit. This and the clustering effect are illustrated on Figure 160.   Of the 24 samples collected in 1970, 13 of them have a reported aluminum concentration of 100 µg/L; the last sample in the data set had a reported aluminum concentration of 0 µg/L. Because of the clustering effect and the fact that all of the most recent data are well below the standard, it seem unlikely that aluminum is really a water quality problem in the ground water mound.

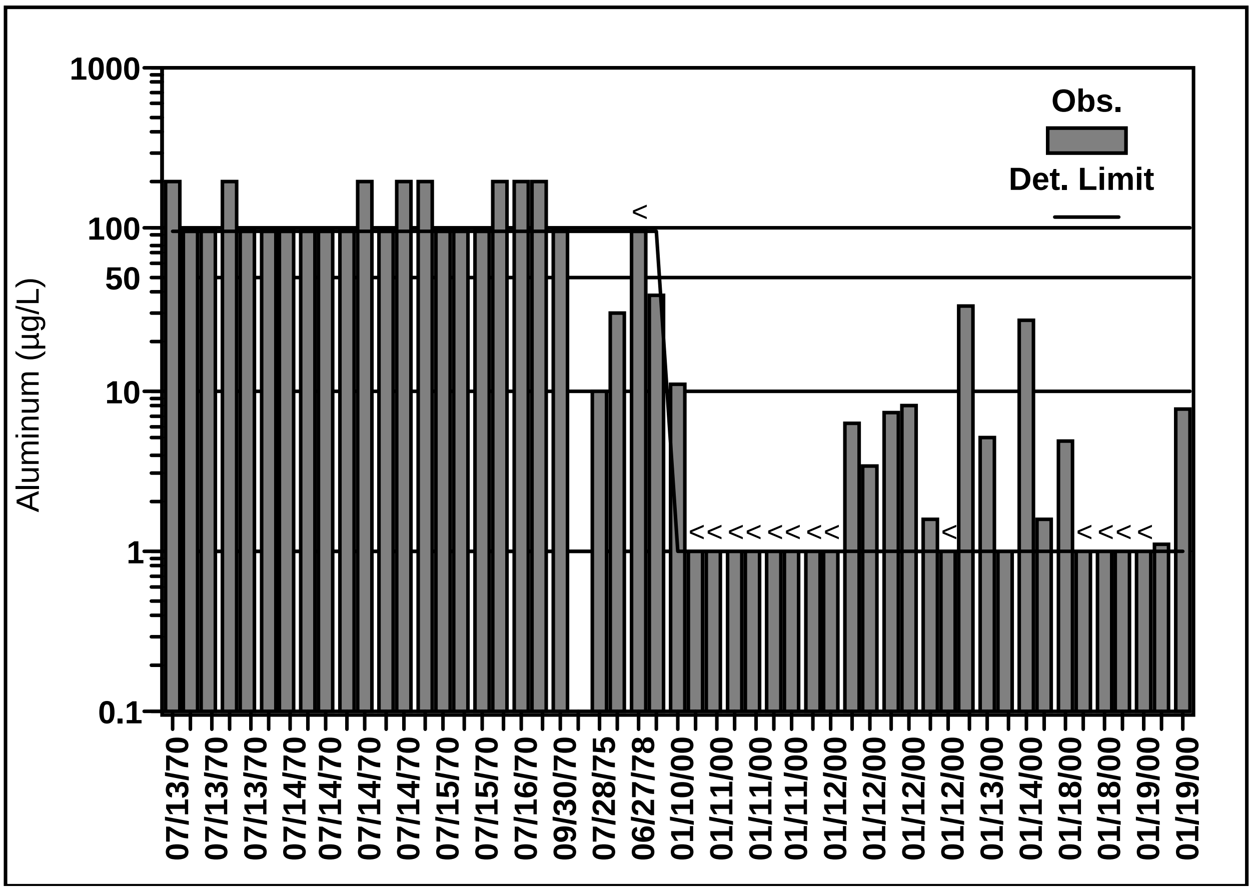


Figure 160. Aluminum data from the 1970's and the recent samples from January 2000

The summary data in Table 50 indicate that the causes of water quality problems were the same as those that were identified in the January 2000 samples, *i.e.* nitrate and selenium. To evaluate this further, the early water quality data were once again compared to the data from the January 2000 samples. The comparison is shown in Table 51. The comparison is based on the Mann-Whitney test as was done in Table 48. Of the 20 variables included in Table 48, 13 show a statistically significant difference between the two sampling periods. Those that do not show a significant difference include temperature, nitrate, boron, chromium, lithium, and selenium. Two of these, nitrate and selenium, are on the list of constituents of concern as noted above. The

Table 51. Comparison of early data (1947-89) with the January 2000 samples from the ground water mound

Variable	Units	1947-89		Jan 2000		Mann-Whitney Test		
		Median	No. of cases	Median	No. of cases	M–W U	Prob.	X^2 (with 1 df)
Temp.	°C	14	180	13.8	27	2813	0.183993	1.765
Sp. Cond.	µS/cm	657	258	840	28	2411	0.003845	8.356
DO	mg/L	6.1	3	0.8	27	71	0.037234	4.340
pH		7.4	218	7.2	28	4106	0.002905	8.866
TDS	mg/L	412	132	N/D[1]	0	—	—	—
NO_3-N	mg/L	2.9	215	3.22	28	2646	0.298115	1.083
Aluminum	µg/L	100	26	1	27	671	< 0.000001	33.641
Arsenic	µg/L	0	12	5	27	33	0.000069	15.847
Barium	µg/L	100	6	134	27	70	0.607496	0.264
Boron	µg/L	80	131	66	27	1710	0.784760	0.075
Chromium	µg/L	0	3	0.7	27	0	0.003795	8.379
Cobalt	µg/L	0	44	0.5	27	25	< 0.000001	61.432
Copper	µg/L	2	25	5.2	27	153	0.000669	11.574
Lead	µg/L	0	44	0.5	27	53	< 0.000001	54.089
Lithium	µg/L	30	25	26	27	340	0.970674	0.001
Nickel	µg/L	0	44	3.5	27	62	< 0.000001	43.734
Selenium	µg/L	6.5	54	4	27	804	0.450838	0.569
Silver	µg/L	0	44	0.5	27	0	< 0.000001	70.000
Vanadium	µg/L	4	16	12	28	76	0.000292	13.121
Zinc	µg/L	10	23	38	27	136	0.000656	11.609

[1] N/D – not determined

fact that there is no significant difference in the concentrations of these constituents indicates that each has been at potentially harmful levels for a considerable period of time.

In addition to aluminum, cobalt, lead, nickel, and silver shows significant differences at a probability level of < 0.000001. Unlike aluminum these trace elements all show a significant increase. In the case of these other trace elements, each shows a median concentration of 0 in the earlier data. This means that over ½ the data points are below the detection limit. Unlike the January 2000 data and some of the earlier observations, the data recorded as 0 are less than some unspecified detection limit. Consequently, the significant increases shown are probably not significant at all. They are based on a comparison of incompatible detection limits. The reporting of the data as being less than some detection limit began in 1975. The detection limit problem also applies to arsenic and chromium, which also show a significant increase, but at a somewhat lower probability level.

The standards shown in tables 50 and 53 as drinking water standards are also Nebraska ground water standards. The primary concern of the Nebraska ground water standards is to protect and improve the quality of ground water for human consumption, agriculture (irrigation and livestock watering), and industry, and other beneficial uses (NDEQ, 1996b). However, it is also noted that in some aquifers the background quality does not meet the standards. For such aquifers the background quality becomes the standard. Consequently, the fact that the drinking water standards shown in tables 3 and 6 are exceeded does not mean that the water exceeds the Nebraska ground water standard. Only the NDEQ can make such a determination.

As was noted above, the purpose of the analysis is to evaluate the water quality of the ground water mound as it relates to options to be considered in the environmental statement for the Platte River endangered species recovery program. For this reason, several of the standards shown in table 47 and 53 are based on aquatic life criteria. There are options to use the ground water mound to either supply water to the river to help meet the flow requirements for the river or to use the water for irrigation supply in exchange for water from Lake McConaughy to meet flow needs. In the former case the aquatic life criteria may apply. In the latter case irrigation standards could apply, although the Nebraska irrigation standards, like the aquatic life criteria, are included in the surface water standards (NDEQ, 1996a). The irrigation standard for selenium is 20 µg/L, which is also its acute (immediately toxic) aquatic life criterion. To complete the list, the drinking water and therefore the ground water standard for selenium is 50 µg/L. The remainder of this report will focus on selenium and to some extent its interrelationships with NO_3–N and specific conductance.

Table 49 showed interrelationships among specific conductance, NO_3-N, and selenium. Updated results are shown in Table 52. The number of data points for the correlation between specific conductance increase from 25 to 214 for the early data, but the correlation coefficient decreased from 0.74 to 0.32 or to less than half of what it had been. However, the significance of the correlation increased somewhat. The same is true of the correlation based on the combined data.

The large increase in the number of NO_3–N observations reflects the use of all of the nitrate forms in the STORET data base. Over the years both the analytical procedure for the nitrate determination and the way nitrate is reported have changed. In the earliest years of the record, nitrate was determined alone. The more recent data are based on a NO_2+NO_3–N determination. In the earliest years, rather than NO_3–N, the data were reported as NO_3. At various times the analysis was for either dissolved or total NO_3–N. All of these forms were considered equivalent, and in the case of the NO_3 data, the values were arithmetically converted to NO_3–N. Since there could be more than one form analyzed on a sampling date, particularly the dissolved and total forms, a preference was established and a nitrate concentration was read into the data set based on the preferences. The order of preference was as follows: dissolved NO_3–N, dissolved NO_3, dissolved NO_2+NO_3–N, total NO_3–N, total NO_3, total NO_2+NO_3–N.

The number of observations in the correlation between selenium and NO_3–N based on the early data more than doubled from 15 to 32 (compare tables 52 and 54). The correlation coefficient (r) decreased from 0.65 to 0.51. However, the statistical significance of the correlation remained below the minimum that could be calculated at 6 decimal places, *i.e.* less than 1 in a million, which was the level set in the statistical package.

Table 52. Correlations among specific conductance, NO_3-N, and selenium based on three different data sets					
Variable 1	Variable 2		Early Data	Jan 2000	Combined
Sp. Cond.	NO_3-N	r	0.322489	0.572610	0.388628
		Prob. > r	0.000001	0.001451	< 0.000001
		n	214	28	242
Selenium	NO_3-N	r	0.818005	0.685388	0.725353
		Prob. > r	< 0.000001	0.000080	< 0.000001
		n	32	27	59
Selenium	Sp. Cond.	r	0.513741	0.469431	0.419595
		Prob. > r	0.000071	0.013498	0.000096
		n	54	27	81

Table 52 also shows the correlations between selenium and specific conductance for each of the 3 data sets. The r-value for the early data decreased from 0.65 to 0.51 and the significance of the relationship increased with the increase of the number of data pairs from 29 to 54. Unlike the other correlations the r-value for the combined data set is lower than those for either of the component data sets. In the cases of the other correlations, the combined data gave an intermediate r-value (Table 52).

It was noted above that there are several different selenium standards. The standards are at concentrations of 5, 20, and 50 µg/L. A comparison of the selenium against standards is shown on Figure 161. Only one sample has exceeded the drinking water/ground water standard of 50 µg/L (Figure 161). Alternatively 8 of the 1969–78 and 3 of the Jan. 2000 samples exceeded the agricultural standard and acute aquatic life criterion. The numbers related to the chronic aquatic life criterion were discussed earlier.

The x-axis on Figure 161 is the township-range-section number. They are arranged by township and range within township, or in other words, generally from south to north. There appears to be an increase in the selenium concentration from south to north in the 1969–78 data, which constitute the majority of the data. This was further evaluated, along with a similar distribution of specific conductance, NO_3–N, boron, and lithium. The latter two were included simply because they had quite a few observations and both are very soluble in water. The results are shown in Table 53.

There is significant variation in specific conductance over both township and range number; however, the interaction between the two is also statistically significant. The interaction indicates that the effect of one variable changes as the other changes. For example, the overall pattern in specific conductance in the four townships indicates that it is lowest in the southernmost township (T5N) and highest in the next adjacent township to the north (T6N); it then drops slightly as one moves farther north (Figure 162). The differences are statistically significant (Table 53).

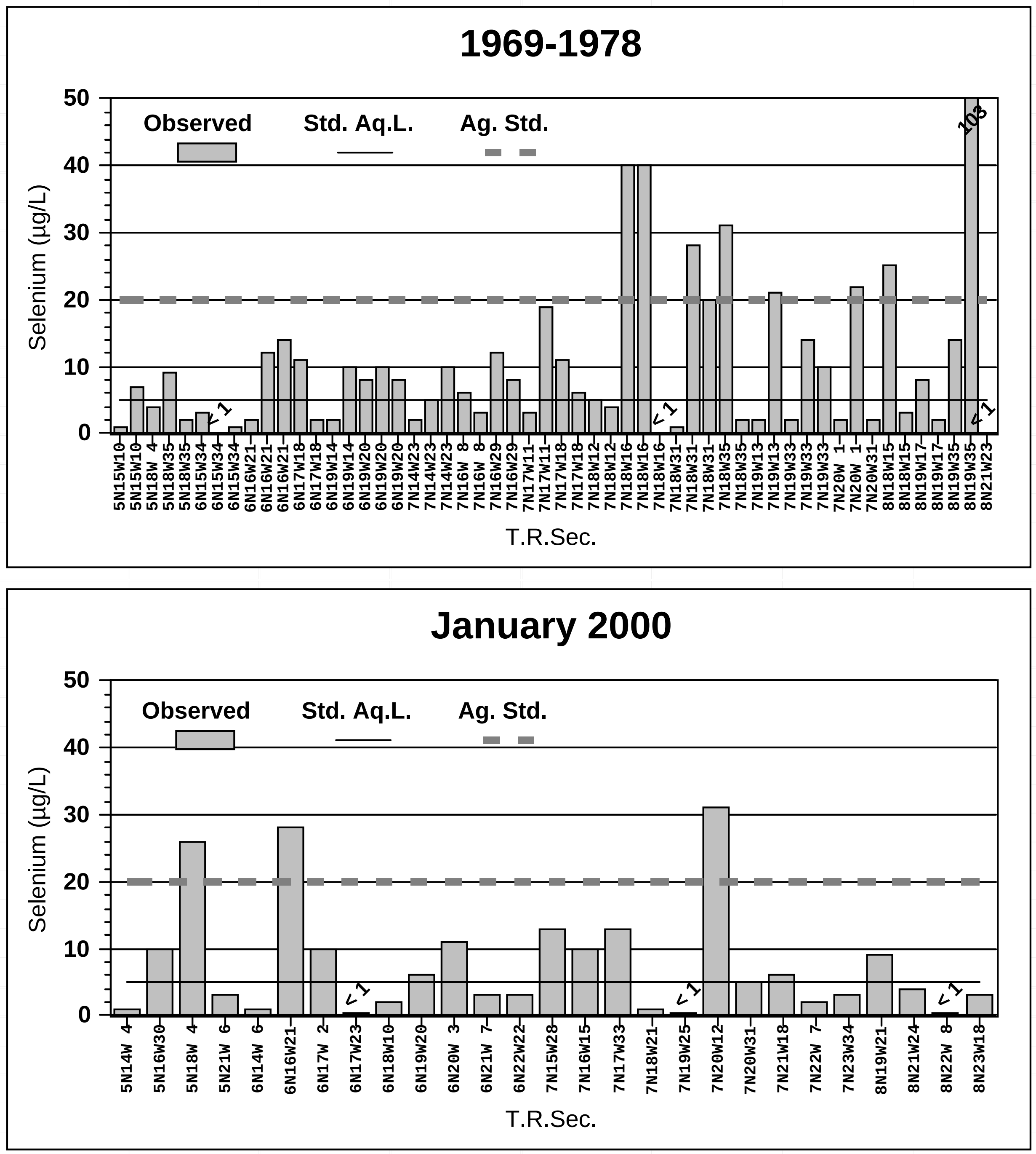


Figure 161. Comparison of selenium in ground water against various water quality standards

Table 53. Analysis of Variance of selected water quality variables over township and range (highlighting flags statistically significant F-ratios)						
Dependent Variable	Source of Variation	df	Mean-Square	F-ratio	Prob. > F	Comments
Sp. Cond.	Township	3	186155	4.747	0.003047	
	Range	4	625722	15.955	< 0.000001	
	Township*Range	12	341464	8.707	< 0.000001	
	Error	266	39217			
NO_3–N	Township	3	99	4.681	0.003425	
	Range	4	55	2.593	0.037434	
	Township*Range	12	27	1.289	0.225938	
	Error	223	21			
Selenium	Township & Range	16	170	0.849	0.627410	Lost df. T = 1, T x R = 2
	Error	64	201			
	Township	3	228	1.174	0.325278	
	Error	77	194			
	Range	4	272	1.421	0.234990	
	Error	76	191			
	Township & County	10	228	1.199	0.306798	Lost df. T = 1
	Error	70	191			
	Township & Range	16	70	0.777	0.704282	Deleted Outlier
	Error	63	91			
Boron	Township	3	13123	4.253	0.006583	
	Range	4	620	0.201	0.937396	
	Township*Range	12	4384	1.421	0.163338	
	Error	138	3086			
Lithium	Township & Range	16	180	1.973	0.046137	Lost df. T x R = 3
	Error	35	91			
	Township	3	113	0.942	0.427688	
	Error	48	120			
	Range	4	428	4.595	0.003257	
	Error	47	93			

The distribution over the ranges from east to west is also shown on Figure 162. The ranges on Figure 162 are paired for the first eight, but the last shown set includes 4 ranges. This is a reflection of the distribution of the data. There are too few observations in some of the township-range combinations to partition the sum of squares without combining the ranges, even in the case of specific conductance which has more observations than any other of the water quality constituents.

Specific conductance shows an increase from the easternmost ranges (13–14W) to the central pair of ranges (17–18W). Specific conductance then declines slightly as one moves farther west (Figure 162). The differences are statistically significant at a probability level below six decimal places (Table 52).

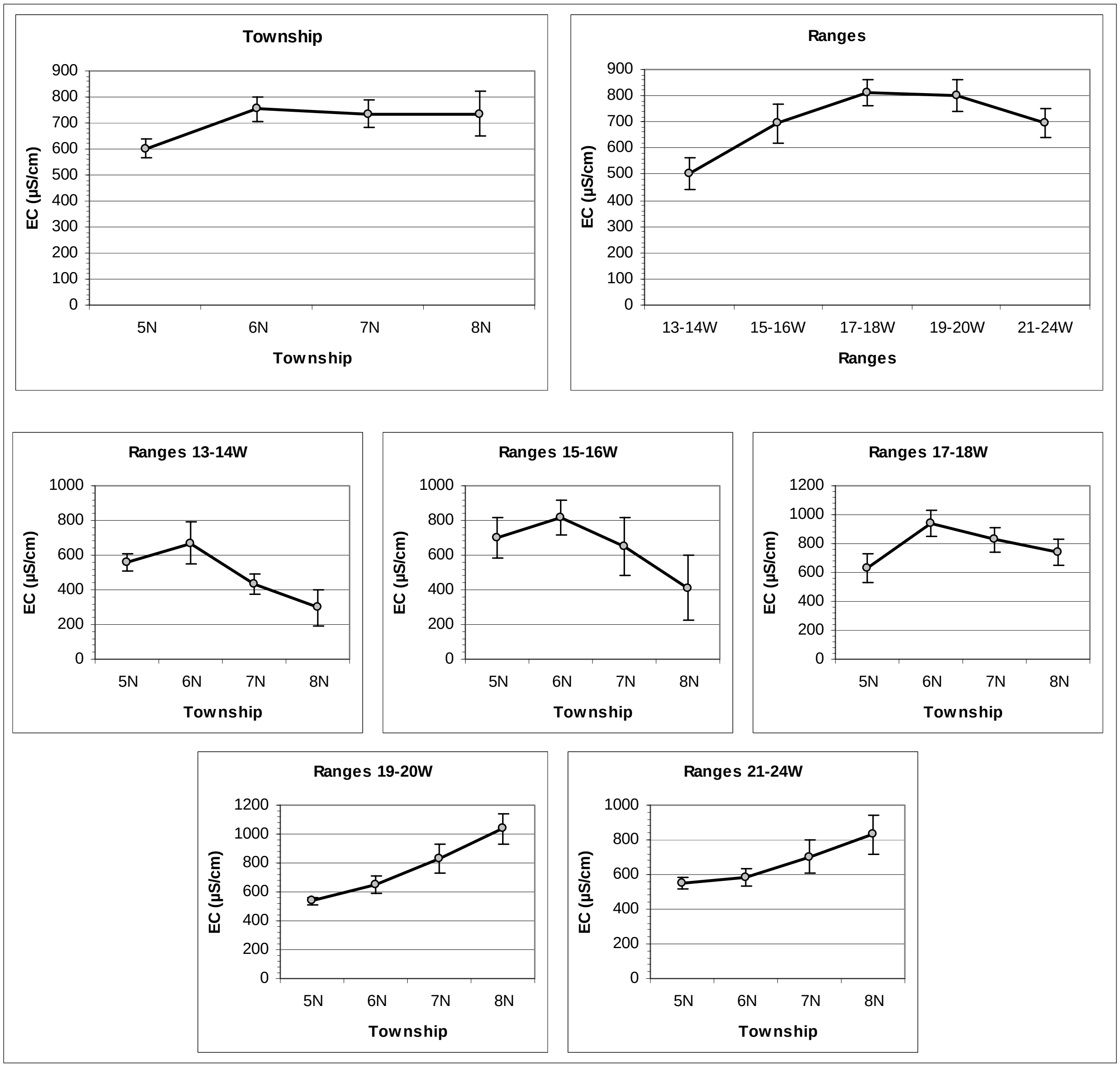


Figure 162. Mean and confidence interval for specific conductance by township and range in the area of the ground water mound

The lower 5 plots on Figure 162 shows the interaction effect. Specific conductance shows a pattern with the townships very similar to the overall pattern in the eastern ranges (13–14W through 17–18W). Although the pattern is similar, the entire distribution is generally shifted upward as one moves to the west. The pattern is completely different in the westernmost two sets of range subdivisions. Specific conductance continually increases to the north in these subdivisions of the ranges, but the mean level of specific conductance increases less in the northernmost townships in ranges 21–24W than in ranges 19–20W (Figure 162). The interaction is more statistically significant than the township effect alone, but not as significant as the ranges effect alone (Table 53).

NO_3–N also shows a statistically significant difference across both townships and ranges, but the interaction is not significant (Table 53). In this case the township effect is more significant than the range effect. In other words, the south to north differences are more pronounced than the east to west differences.

There are a number of sets of ANOVA output summarized in Table 53. The first shows the results of the ANOVA for the township-range evaluation with interactions. As noted in the comments, the ANOVA lost degrees of freedom (df), which means that some of the cells were empty. This is illustrated in Figure 163. There are no data for township 8N in the easternmost two sets of ranges (13–14W and 15–16W), and there are no data for T5N in ranges 19–20W. Consequently, the interactions could not be evaluated, and the township and range effects were evaluated independently using oneway ANOVA. However, the overall ANOVA for the township and range effect was not statistically significant (Table 53).

The ANOVA for the township effect is not statistically significant (Table 53). The plots of the township effect is shown on Figure 163 and indicates that there is generally an increase from south to north. However, the error bars show that there is a very large amount of variation about the mean. Because of the overlap in the error bars, the differences in the mean selenium concentrations are not statistically significant.

The range effect is similarly not statistically significant (Table 53). The plot of the ranges effect shows that the mean selenium concentration is generally higher in the ranges in the center of the mound, but once again the error bars are large (Figure 163). The results is that the differences are once again not statistically significant.

An attempt was made to look at the selenium distribution by further consolidating the range data into counties. The three counties in the mound area are each 4 ranges wide. However, because there were no data for T8N on either of the easternmost two sets of ranges, there are no data for T8N in Kearney; those ranges are equivalent to Kearney County. The selenium still showed no significant difference on the basis of county either (Table 53).

The last selenium ANOVA in Table 53 was based on the same township and range setup as the first, but the very large outlier (103 µg/L of selenium) was deleted from the data set. As was noted above the error bars are very large on many of the subsets of selenium data. However, the effect of deleting the outlier was to further decrease the probability associated with the F-ratio. Consequently the significance of the ANOVA declined. It should be noted that a nonparametric ANOVA, which would not be affected by outliers also showed no statistically significant difference in selenium between either township or range.

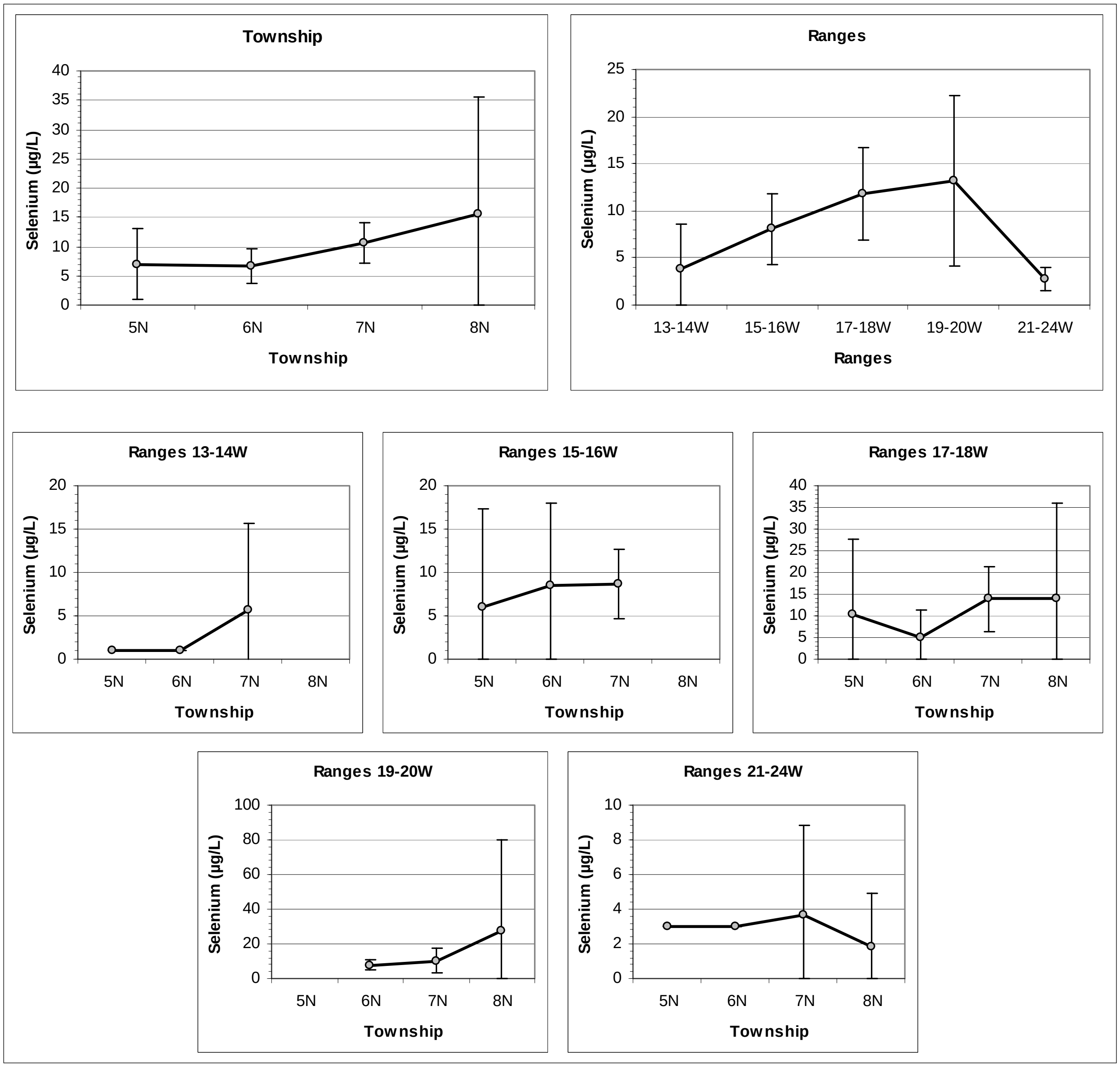


Figure 163. Mean and confidence interval for selenium by township and range in the area of the ground water mound

The last two sets of ANOVA's in Table 53 are for boron and lithium. Boron is included as a curiosity, and only shows a statistically significant difference with township. Neither the range nor the interaction is significant. Lithium is included to illustrate a point about the first selenium ANOVA, which was not significant. In the case of lithium the overall ANOVA is significant even though the interaction could not be run. This ANOVA represents the effects of a lumped statistical model for township and range. This lumped lithium ANOVA is statistically significant. To further evaluate this, the equivalent oneway ANOVA's are also shown in Table 53. These show that the statistical significance of the lumped model is due to the difference across ranges. There is no significant difference across townships. In other words there is no difference in lithium from south to north, but there is from east to west. This latter difference is responsible for the significance of the lumped model. In the case of selenium, neither of the oneway models was significant, and consequently, the lumped model was not significant either. The three multivariate selenium ANOVA's consist of lumped models.

Herbicides in ground water

Table 47 summarizes the data from samples collected during January 2000. The last block in the table shows a summary of the samples for total triazine herbicides. Atrazine has been identified as being the most likely herbicide to exceed its MCL in the Platte River alluvial aquifer (Frenzel *et al.*, 1998); cyanazine, another triazine herbicide, was also identified as a potential ground water contaminant. The January 2000 samples did not include a breakdown of individual herbicides. However, a summary of data for several triazine herbicides is included in Table 1. These include atrazine, cyanazine, propazine, and simazine. According to EPA (2003), there are drinking water MCL's for both atrazine (3 µg/L) and simazine (4 µg/L). All four of these herbicides are listed as possible human carcinogens and health advisories (HA) have been developed. The adult lifetime HA's for atrazine and simazine are their MCL's; for cyanazine and propazine, the HA's are 1 µg/L and 10 µg/L respectively. The NDEQ (1996a) also has a 3 µg/L public water supply criterion for atrazine.

An attempt was made to retrieve the individual records for the herbicide data summarized in Table 45 from EPA's STORET data base. For atrazine, cyanazine, and simazine, a greater number of samples than previously shown in Table 45 were retrieved. There were no data for propazine, although the summary data indicated that there were 16 samples in the NRC data base. The retrieved herbicide data, along with temperature, specific conductance, and various measures of nitrate, are summarized in Table 54.

Table 54. Summary of Physical, Nitrate, and Triazine Data for the Ground Water Mound

	Period of Record	Water Temperature (°C)	Conductivity at 25°C (µmho/cm)	NO_3-N Total (mg/L)	NO_2 & NO_3 N - total (mg/L)
Minimum	03/30/45	10	170	0.0	0.01
Median	—	14	657	2.7	4.25
Maximum	08/10/89	24	1970	47	17
No. of Obs.	266	180	258	61	44
No. > D.L.	—	—	—	61	43
	NO_2 & NO_3 N - diss. (mg/L)	**Simazine Total (µg/L)**	**Atrazine Total (µg/L)**	**Cyanazine Total (µg/L)**	**Total Triazines (µg/L)**
Minimum	0.0	< 0.05	< 0.04	< 0.10	< 0.10
Median	2.9	< 0.05	< 0.10	< 0.10	< 0.10
Maximum	14	0.10	0.98	< 0.40	1.03
No. of Obs.	83	48	48	39	48
No. > D.L.	83	2	18	0	19

There period of record for the STORET data is shown as including the years of 1945 through 1989 in the Tri-County area that encompasses the ground water mound. The retrieved herbicide data included USGS data collected between 1984 and 1988. There was no detectable cyanazine in any of the samples retrieved (Table 54). Only two of the samples showed detectable simazine. Eighteen of the samples (more than ⅓) showed detectable atrazine. None of the samples exceeded the individual MCL's or HA's for the respective triazine herbicides. All of the atrazine samples were also below the Nebraska aquatic life criterion.

Based on the 1984-88 data, the total triazines were dominated by atrazine. Total triazines were calculated from the data for the individual herbicides. In all but two cases the total triazines were based solely on the atrazine concentration.

The total triazine calculation was based only measured concentrations that were greater than the detection limit. Since all of the cyanazine analyses were below the detection limit, the cyanazine data were not entered into the calculation. The total was calculated based only on the atrazine and simazine concentrations, as follows. If both herbicides were below the detection limit, the total triazine concentration was set to 0.2 µg/L, which was approximately ½ the detection limit in the January 2000 samples. If the one or the other of the two triazines were greater than their respective detection limits, the total triazine concentration was set to that concentration. If both were present at detectable concentrations, the two were summed; this only happened once in the 1984-88 data set. The total triazine concentrations were compared to see if the concentrations were different between the two time periods. The comparison appears in Table 55. Data for specific conductance and nitrate-nitrogen are also compared in the table; each of the inorganic water quality constituents shows no significant difference between the two time periods. Alternatively there is a significant difference in total triazines (Table 55), although just barely so. The difference in triazines shown in Table 55 is an increase; however, because of the effect of detection limits, this may or may not be the case.

Table 55. Comparison of total triazines between the 1980's and January 2000

Variable		1984-88	Jan. 2000
Conductivity at 25°C	Median (µmho/cm)	725	820
	X^2	1.4417	
	Probability > X^2	0.2299	
Nitrate	Median (mg/L)	3.1	2.76
	X^2	0.0004	
	Probability > X^2	0.9842	
Total Triazines	Median (µg/L)	0.02	0.08
	X^2	4.0711	
	Probability > X^2	0.0436	

The detection limits for the individual triazine herbicides in the 1984-88 data set are as follows: atrazine, 0.05-0.1 µg/L; cyanazine, 0.1-0.4 µg/L; and simazine, 0.05-0.1 µg/L. To a great extent the number of samples greater than the detection limit is proportional to the magnitude of the detection limit. The detection limit for the Jan. 2000 total triazine samples was 0.05 µg/L, which is very close to the lowest detection limit for any one of the triazines in the 1984-88 samples. To compensate for this, the data were compared to a detection limit of 0.1 µg/L. The results of the comparison are shown in Table 56, which includes the number of samples greater than 0.1 µg/L in each of the data sets along with the total number of samples in the data set. There are about an equal number of samples with detectable triazines in the two data sets; however, there are nearly twice as many samples in the 1980's data set. The comparative difference is that about ¼ of the samples have detectable triazines in the 1980's data set and about ½ have detectable triazines in the January 2000 data set. This would indicate that there has been an increase in triazines in the ground water under the mound since the 1980's.

Table 56. Comparison of total triazine data to common detection limit

Samples	Total > 0.1µg/L	Total samples
1984-88	13	48
Jan. 2000	14	29

The maximum atrazine in the ground water in the 1980's data set was 0.98 µg/L. This is very near the Nebraska aquatic life criterion. If there has been a generalized increase in atrazine, it may soon exceed the criterion with regularity in the future. If this were to occur, atrazine would be a cause for caution in the operation of any ground water recharge system in the area of the mound.

Water quality in small streams draining the ground water mound

The USGS has been studying the water quality of various tributaries to the Platte River that drain the ground water mound. The data were provided to the EIS team by Rich Holloway of the Tri-Basin Natural Resource District, Holdrege, Nebraska (personal communication of February 17, 2000, to Jim Yahnke, Hydrologist, Bureau of Reclamation, Denver, Colorado). The data include a wide range of nutrient (nitrogen and phosphorus) and pesticide concentrations. There were also flow, specific conductance, water temperature, dissolved oxygen, and pH measurements in the data set. The specific conductance and pesticide data, along with water quality standards for aquatic life or agriculture (specific conductance), are summarized in Table 57. Each of the tributaries are classified for support of warmwater aquatic life and agriculture, just like the mainstem of the Platte River.

The vast majority of the pesticide concentrations are below their respective detection limits (Table 57). The only pesticides for which the majority of the samples were above detectable concentrations were atrazine, two of its degradation products, and metolachlor. There are no water quality standards for the atrazine compounds, but only for atrazine and metolachlor. Metolachlor, like all of the other pesticides for which there are ground water standards, did not exceed the standard (Table 57). Atrazine exceeded the standard in nearly ½ of the samples collected in North Dry Creek and the Fort Kearney Slough. As has been noted before, atrazine is a recognized widespread water quality problem in Nebraska (NDEQ, 1996c).

The discussion of triazines indicated that atrazine in the ground water during the 1980's was approaching the water quality standard. The underlying assumption is that the tributaries are predominantly ground water that is draining from the mound area. If this is true, the 1997-98 data for the tributaries indicate that the ground water exceeds the standard for atrazine regularly. However, before such a conclusion can be drawn, the question of the source of the tributaries should be addressed.

The specific conductance of the various tributaries shows quite a bit of variation (Table 57). The median specific conductance for the tributaries ranges from less than 500 to over 1100 µS/cm. The median specific conductance of the January 2000 ground water samples was 820 µS/cm (Table 47). The specific conductance of the ground water varied greatly over the study area in the January 2000 samples. It would be difficult to say that any surface water was greatly different from the ground water in the mound no matter what the specific conductance readings were. This was further evaluated by selecting wells located in the township-range combinations within the drainages of each of the tributaries. The results are shown in Table 58, which once again includes the median specific conductance of the tributaries and the ground water. The median ground water specific conductance includes both the earlier and the more recent data; the median is much lower than the one shown for the January 2000 data, *i.e.* 669 and 820 µS/cm respectively.

When the specific conductance of the tributaries is compared to that of the local ground water, there are still some significant differences (Table 58), including those for North Dry Creek, Whisky Slough, and the Downstream Drain. Three of the tributaries, Whiskey Slough, Ft, Kearney Slough, and the Downstream Drain, have relatively low specific conductance (460-527 µS/cm), but in the case of the latter two tributaries, it is still higher than the adjacent ground water (449 µS/cm). There is no statistically significant difference in their specific conductance based on the Kruskal-Wallis ANOVA ($X^2 = 4.5$, probability • 0.1). Each of the three tributaries runs generally parallel to the Platte River and is located very near to the river. The tributaries and the adjacent ground water may be more affected by the Platte River, particularly it high flows, than by the ground water mound.

Table 58. Comparison of the specific conductance (µS/cm or µmho/cm) of the ground water (GW) and that of the tributaries (SW)

Data set	Median GW	Median SW	X^2	Prob. > X^2
All	669	881	22.25	0.000002
Plum Creek	795	892	1.94	0.163187
N Dry Creek	793	970	14.67	0.000128
Whisky Slough	793	461	15.15	0.000099
Ft. Kearney Slough	449	490	2.63	0.104533
Downstream Drain	449	527	4.56	0.032658

To further confuse matters, there appears to be a difference in conditions over time. Table 59 shows a comparison of the flow and specific conductance data for North Dry Creek over time. The median flow in the creek has more than doubled between the late-1960's/early 1970's to the 1990's. This could be a reflection of the fact that the 1990's have been a much above average period in terms of moisture; however, it could also be due to the rise in ground water in the mound. Table 59 also shows that the specific conductance has increased significantly since the late-1960's/early 1970's. The data are plotted on Figure 164. The plot shows that the specific conductance data in the late-1960's/early-1970's mostly remained below 1,000 µmho/cm, while the later data were practically all above 1,000 µmho/cm. If the increased flow were due to precipitation, which is extremely low in dissolved solids, the specific conductance would be expected to decline, rather than increase. The increase in specific conductance in conjunction with the increase in flow, indicates that the flow increase is most likely due to the rise in ground water in the mound.

Table 57. Summary of pesticide data from streams draining the ground water mound

		Specific Conductance (µS/cm)	Propachlor Diss, (µg/L)	Butylate Diss, (µg/L)	Simazine Diss, (µg/L)	Prometryn Diss, (µg/L)	Prometon Diss, (µg/L)	Deisopropyl Atrazine Diss, (µg/L)	Deethyl Atrazine Diss, (µg/L)	Cyanazine Diss, (µg/L)
Aquatic Life Standard		2000	8	—	—	—	—	—	—	—
Plum Creek near Smithfield	No. > Std.	0	0	—	—	—	—	—	—	—
	No. > D.L.	17	1	0	0	0	1	5	13	5
	No. of Obs.	17	22	4	22	18	22	18	22	22
	Median	892	<.0500	<.0020	<.0500	<.0500	<.0500	<.0500	0.009	<.0500
	Maximum	978	<.0500	<.0020	<.0500	<.0500	<.0500	0.220	0.230	0.200
Whisky Slough 1mi E of Phelps-Kearney Co Line	No. > Std.	0	0	—	—	—	—	—	—	—
	No. > D.L.	13	0	0	2	0	0	13	16	0
	No. of Obs.	13	20	5	20	15	20	15	20	20
	Median	461	<.0500	<.0020	<.0500	<.0500	<.0500	0.140	1.490	<.0500
	Maximum	573	<.0500	<.0020	0.003	<.0500	<.0500	0.580	2.340	<.0500
North Dry Cr 2.0 Mi SW of Bridge South of Kearney	No. > Std.	0	0	—	—	—	—	—	—	—
	No. > D.L.	22	4	0	9	0	12	18	28	12
	No. of Obs.	22	30	8	30	22	30	22	30	30
	Median	1145	<.0500	<.0020	<.0500	<.0500	<.0500	0.150	0.460	<.0500
	Maximum	1730	0.080	<.0020	0.051	<.0500	0.166	1.980	4.030	4.320
Fort Kearney Slough near Newark	No. > Std.	0	0	—	—	—	—	—	—	—
	No. > D.L.	19	1	0	5	0	3	21	27	0
	No. of Obs.	19	29	6	29	23	29	23	29	29
	Median	490	<.0500	<.0020	<.0500	<.0500	<.0500	0.340	1.230	<.0500
	Maximum	1030	0.120	<.0020	0.140	<5.00	0.023	2.010	5.380	<.0500
Downstream Drain near Newark	No. > Std.	0	0	—	—	—	—	—	—	—
	No. > D.L.	10	2	2	6	0	5	12	19	3
	No. of Obs.	10	19	7	19	12	19	12	19	19
	Median	527	<.0500	<.0020	<.0500	<.0500	<.0500	0.200	1.320	<.0500
	Maximum	850	0.136	0.114	0.126	<.0500	0.106	0.320	3.480	0.162

Table 57. (continued)

		Fonofos Diss, (µg/L)	Alpha BHC Diss, (µg/L)	P,P' DDE Diss, (µg/L)	Ametryn Diss, (µg/L)	Propazine Diss, (µg/L)	Terbutryn Diss, (µg/L)	Chlorpyrifos Diss, (µg/L)	Lindane Diss, (µg/L)	Dieldrin Diss, (µg/L)
Aquatic Life Standard		—	0.131	0.0084	—	—	—	0.041	0.08	0.00144
Plum Creek near Smithfield	No. > Std.	—	0	0	—	—	—	0	0	0
	No. > D.L.	0	0	0	0	0	0	1	0	0
	No. of Obs.	4	4	4	18	18	18	4	4	4
	Median	<.0030	<.0020	<.0060	<.050	<.050	<.05	<.0040	<.004	<.001
	Maximum	<.0030	<.0020	<.0060	<.050	<.050	<.05	0.007	<.004	<.001
Whisky Slough 1mi E of Phelps-Kearney Co Line	No. > Std.	—	0	0	—	—	—	0	0	0
	No. > D.L.	0	0	1	0	1	0	0	0	0
	No. of Obs.	5	5	5	20	20	20	5	5	5
	Median	<.0030	<.0020	<.0060	<.050	<.050	<.05	<.0040	<.004	<.001
	Maximum	<.0030	<.0020	0.001	<.050	0.080	<.05	<.0040	<.004	<.001
North Dry Cr 2.0 Mi SW of Bridge South of Kearney	No. > Std.	—	0	0	—	—	—	0	0	0
	No. > D.L.	1	0	0	0	8	0	3	1	0
	No. of Obs.	8	8	8	22	22	22	8	8	8
	Median	<.0030	<.0020	<.0060	<.050	<.050	<.05	<.0040	<.004	<.001
	Maximum	0.003	<.0020	<.0060	<.050	0.310	<.05	0.015	0.002	<.001
Fort Kearney Slough near Newark	No. > Std.	—	0	0	—	—	—	0	0	0
	No. > D.L.	0	0	0	6	5	0	2	0	3
	No. of Obs.	6	6	6	0	23	23	6	6	6
	Median	<.0030	<.0020	<.0060	0.100	0.060	<.05	<.0040	<.004	<.001
	Maximum	<.0300	<.0020	<.0060	0.200	0.710	<5.0	0.033	<.004	0.005
Downstream Drain near Newark	No. > Std.	—	2	2	—	—	—	2	2	0
	No. > D.L.	2	2	3	0	0	0	3	2	2
	No. of Obs.	7	7	7	12	12	12	7	7	7
	Median	<.0030	<.0020	<.0060	<.050	<.050	<.05	<.0040	<.004	<.001
	Maximum	0.105	0.133	0.067	<.050	<.050	<.05	0.101	0.143	0.107

Table 57. (continued)

		Metolachlor Diss, (µg/L)	Malathion Diss, (µg/L)	Parathion, Diss, (µg/L)	Diazinon, Diss, (µg/L)	Atrazine Diss, (µg/L)	Alachlor Diss, (µg/L)	Acetochlor Diss, (µg/L)
Aquatic Life Standard		100	0.1	0.013	—	1	76	—
Plum Creek near Smithfield	No. > Std.	0	0	0	—	0	0	—
	No. > D.L.	12	0	0	0	16	2	2
	No. of Obs.	22	4	4	4	22	22	22
	Median	0.210	<.005	<.004	<.002	0.110	<.050	<.0500
	Maximum	0.217	<.005	<.004	<.002	0.680	0.009	0.010
Whisky Slough 1mi E of Phelps-Kearney Co Line	No. > Std.	0	0	0	—	1	0	—
	No. > D.L.	10	0	0	0	16	3	2
	No. of Obs.	20	5	5	5	20	20	20
	Median	<.050	<.005	<.004	<.002	0.584	<.050	<.0500
	Maximum	5.590	<.005	<.004	<.002	7.720	0.240	0.060
North Dry Cr 2.0 Mi SW of Bridge South of Kearney	No. > Std.	0	0	0	—	14	0	—
	No. > D.L.	22	1	0	0	30	10	9
	No. of Obs.	30	8	8	8	30	30	30
	Median	0.240	<.005	<.004	<.002	0.860	<.050	<.0500
	Maximum	7.600	0.007	<.004	<.002	15.800	0.151	5.570
Fort Kearney Slough near Newark	No. > Std.	0	0	0	—	12	0	—
	No. > D.L.	18	0	1	0	27	2	2
	No. of Obs.	23	6	6	6	27	29	29
	Median	2.870	<.005	<.004	<.002	6.550	<.050	<.0500
	Maximum	12.200	<.005	0.020	<.002	63.400	0.210	0.170
Downstream Drain near Newark	No. > Std.	0	0	0	—	1	0	—
	No. > D.L.	11	2	3	2	19	4	2
	No. of Obs.	19	7	7	7	19	19	19
	Median	0.006	<.005	<.004	<.002	0.449	<.050	<.0500
	Maximum	0.454	0.135	0.101	0.107	1.070	0.121	0.125

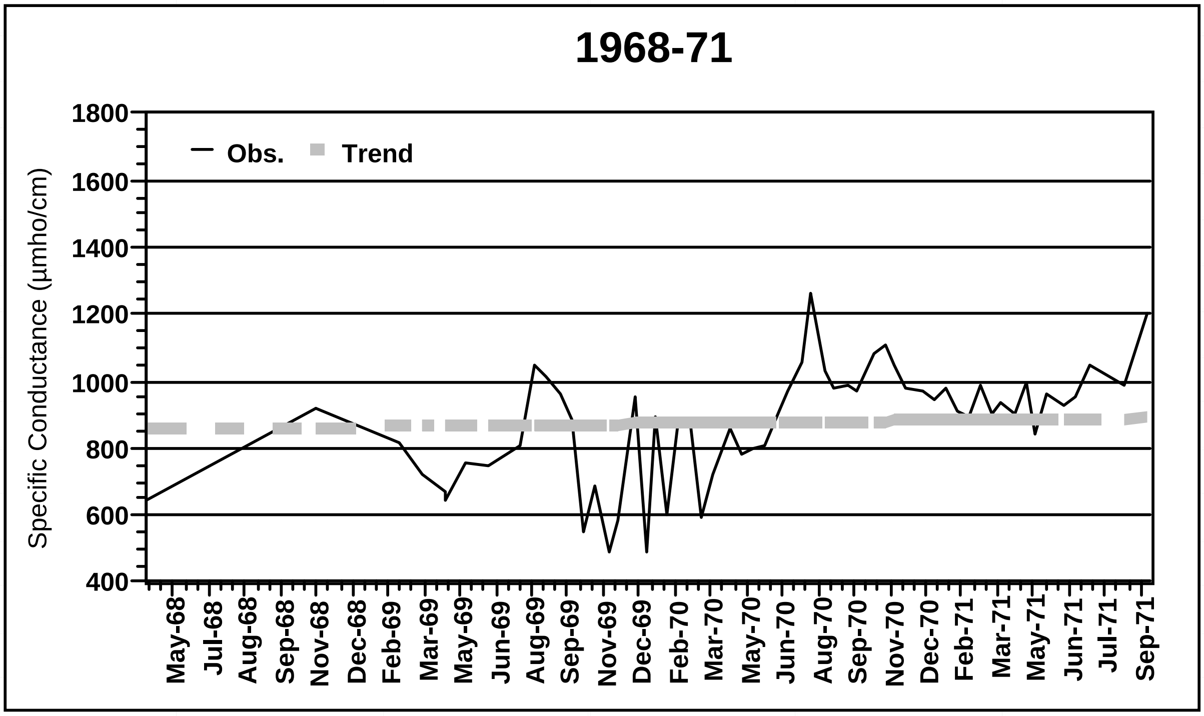
Table 59. Comparison of early *vs*. recent flow and specific conductance data: A. t-test for the independence of means; B. trend analysis				
A.			t-test	
Variable	Ave. 1968-71	Ave. 1994-98	t	Prob. > t
Flow (ft^3/s)	10.3	23.8	4.13	0.000000028
EC (µmho/cm)	878	1139	9.03	< 0.000000001
B. Regression on Julian Date				
Variable	r	Prob. > r	b_1	b_0
Flow	0.5998	< 0.000001	0.001440	-26.78
EC	0.6321	< 0.000001	0.026857	186.18

The ground water quality problem that could have adverse biological effects is the one with selenium. There has been only one selenium sample collected from North Dry Creek, and none from the other tributaries. It was noted earlier that there was a highly significant correlation between NO_3–N and selenium. In the absence of much selenium data, it may be possible to at least screen the tributary data for selenium based on the NO_3–N data, based on the assumption that the tributary flows are predominantly ground water. This is done on Figure 165.

Table 60 summarizes the nitrogen data broken down by chemical species. In addition to the NO_3–N data, the table includes data on NH_4-N. The NDEQ (1996c) has listed the Middle Platte as not supporting the aquatic life beneficial use because of violations of the NH_4-N standard. The NH_4-N standard varies with pH and temperature; so specific numbers are not given. The exact location is not specified. An evaluation of the NH_4-N data for the mainstem of the Middle Platte indicated that the standard was met there. This means that the nonattainment appears to be confined to tributaries. Early data indicate that one sample also exceeded the NH_4-N standard in Ft. Kearney Slough, but this would not likely be a cause for noncompliance. However, the standard was exceeded in about 27 percent of the samples collected during 1996-98 from North Dry Creek. This would seem to be either the basis for noncompliance or at least one of the contributing sites.

The Nebraska ground water standard, as well as the drinking water MCL, for NO_2-N is 1.0 mg/L. The NO_2-N standard was exceeded only by the maximum concentrations in Whisky slough and Ft. Kearney Slough (Table 60). NO_2-N is usually considered a transient species formed during the chemical reduction of NO_3-N to NH_4-N. The presence of significant amounts of NO_2-N in surface water is an indicator of the influence of ground water in the stream.

The next two columns of Table 60 are devoted organic plus ammonium concentrations. They are included primarily for completeness. The data do indicate the presence of organic matter, which is an indicator of surface water influences. Consequently, the streams are not just an indicator of ground water quality.



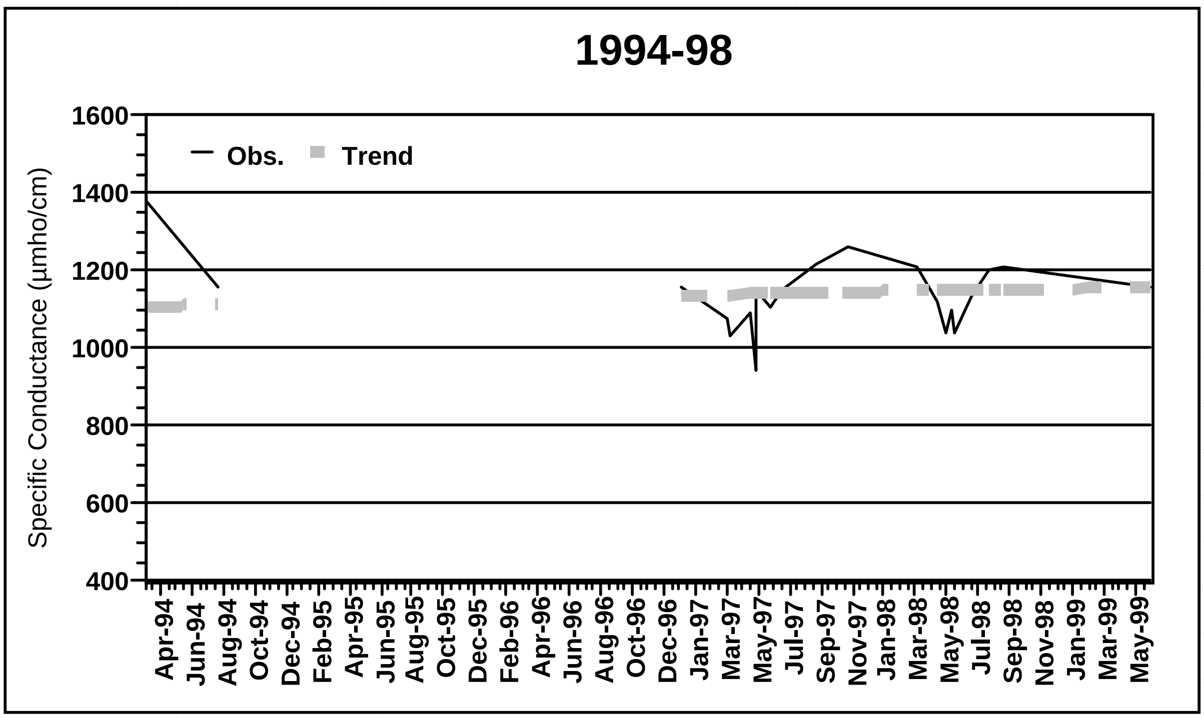


Figure 164. Trend in specific conductance in North Dry Creek since the late-1960's

Table 60. Summary of nitrogen speciation data

Station Name		Ammonia NH_4-N (mg/L)	Nitrite NO_2-N (mg/L)	Organic + NH_4-N Diss. (mg/L)	Organic + NH_4-N Total (mg/L)	NO_2+ NO_3–N (mg/L)
Plum Creek near Smithfield	Minimum	<.015	<.010	0.20	0.43	0.1
	Median	0.058	0.027	0.37	0.73	1.4
	Maximum	0.502	0.131	1.20	2.10	1.9
	No. of Obs.	16	16	16	16	16
	No. > Std.	0	0	No Std.	No Std.	0
Whisky Slough 1 mi. E of Phelps-Kearney Co Line	Minimum	<.015	<.010	<.10	<.20	<.050
	Median	0.045	0.145	0.40	0.46	13.3
	Maximum	0.622	1.07	1.70	1.80	17.7
	No. of Obs.	15	15	15	15	15
	No. > Std.	0	1	No Std.	No Std.	11
North Dry Cr 2 mi. SW of bridge S of Kearney	Minimum	<.015	0.048	<.10	0.55	1.7
	Median	0.052	0.1385	0.62	1.10	6.8
	Maximum	5.84	0.619	3.40	11.00	9.8
	No. of Obs.	22	22	22	22	22
	No. > Std.	6	0	No Std.	No Std.	0
Fort Kearney Slough near Newark	Minimum	<.015	<.010	<.20	<.20	<.050
	Median	0.045	0.092	0.59	0.62	16.3
	Maximum	3.69	3.65	15.00	29.00	23.9
	No. of Obs.	19	19	19	19	19
	No. > Std.	1	1	No Std.	No Std.	16
Downstream Drain near Newark	Minimum	<.015	0.023	<.20	0.21	11.4
	Median	0	0.046	0.38	0.42	20.1
	Maximum	0.051	0.116	1.30	2.00	23.3
	No. of Obs.	11	11	11	11	11
	No. > Std.	0	0	No Std.	No Std.	11

NO_3-N exceeds its standard in a large percentage of the samples from three of the five streams shown in Table 60. North Dry Creek is second lowest in NO_3-N and did not exceed the standard in any of the samples.

The ground water component of stream flow is known as base flow. The base flow varies much less than other components of flow, at least on a daily basis. Water quality samples collected during base flow should be representative of the ground water quality in the aquifer contributing the flow. There are full-time gages located at each of the sites shown in Table 60. The daily flow data for each of the gages was retrieved from the USGS National Water Information System. The data were plotted and periods of base flow were flagged from recession curves on the hydrographs for each site. Water quality samples collected during these periods of base flow were extracted from each data set.

The ground water NO_3-N/selenium regressions were used to project a selenium concentration for the base flow periods in North Dry Creek. The regression derived from the pre-1990 data was used to estimate a selenium concentration for the data from 1969 through 1971; the regression derived from the pre-1990 plus the January 2000 data was used to develop an estimated selenium concentration for the 1997 and 1998 data. The base flow NO_3-N and the associated predicted selenium concentrations are shown on Figure 165. The NO_3-N ranged from 0.3 to 8.1 mg/L as nitrogen; as noted above, none of the NO_3-N samples exceeded the water quality standard. The predicted selenium ranged from 7 to 21 µg/L; in this case even the minimum would be greater than the aquatic life criterion for selenium.

The EPA collected one water sample from North Dry Creek in June 1994 that was analyzed for selenium, although not for NO_3-N. The sample showed 9.6 µg/L of selenium, which is within the estimates from the ground water NO_3-N/selenium regressions. Based on this, the ground water NO_3-N/selenium regressions predict a selenium concentration that at least bracket the one measured concentration in the creek. However, the regression estimate also indicates that the aquatic life criterion for selenium, 5 µg/L, is probably exceeded frequently in North Dry Creek, but there are no data to confirm this result.

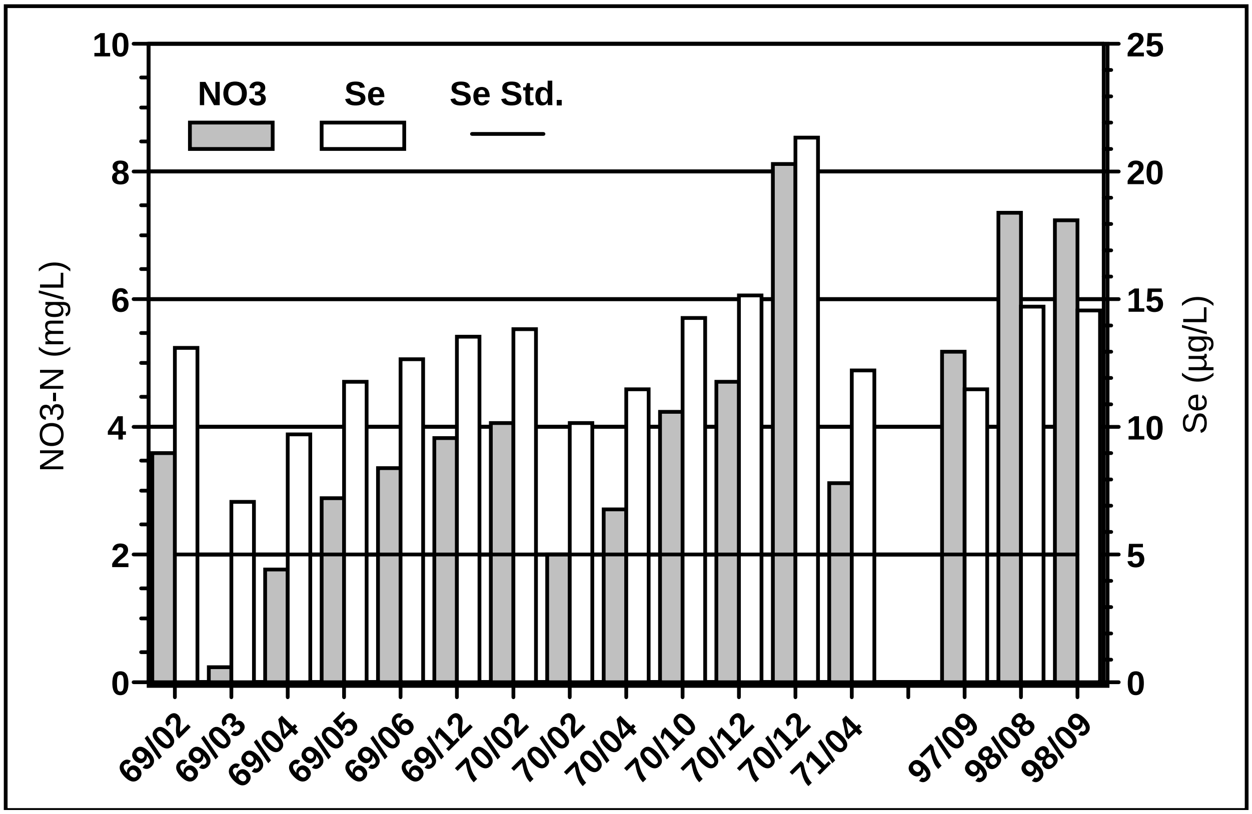


Figure 165. Baseflow nitrate-nitrogen (NO3-N) and predicted selenium in North Dry Creek

Potential effects of ground water management plans

A number of plans currently being considered for water to augment the flow of the Platte River involve ground water management. These plans include both direct and indirect augmentation of the river with ground water. The plans currently under consideration are summarized in Table 61. This section of the report will evaluate these plans from a water quality perspective.

The first plan in Table 61 was developed by Reclamation (USBR). In this plan water would be stored in the area of the ground water mound. The plan envisions using water from the mound to irrigate lands as shown in Table 61 in exchange for water from Lake McConaughy. The water that is used would be replaced by diverting excess flows during periods of low demand for water. The ground water would not be delivered directly to the Platte River.

Figures 158 and 159 show that there are areas that are high in selenium within the source areas described in Table 61 as being targeted for the ground water management plan, but there does appear to be an area of low selenium in the January 2000 samples that may be a suitable source. Potential recharge areas contemplated in the Reclamation plan are shown on Figure 166. There are several samples showing high concentrations of selenium in the potential recharge area located to the west.

Table 61. Summary of management plans for the ground water mound area				
Plan	Water Source	Quantity	Receptor	Comments
USBR	20,000 acre area 6 to 12 miles north of Holdrege and/or 20,000 acre 3 to 11 miles west of Minden	25,000 acre feet	Farm fields in the same area or a short distance farther east.	Water pumped from 30 to 50 feet depth. Replaced each year from river flows.
Boyle option 1	Along the Funk-Odessa road (18 wells) & along A-33.5 Lateral between Funk and Axtel (23 wells)	1,400 to 3,800 acre feet	Platte River via North Dry Creek, and Lost Creek.	Much like USBR except it is smaller in scope and the area is better defined. May milk the canals
Boyle option 2	Ground water (15% " would not go to mound storage.)	1,400 to 3,800 acre feet	none	Water exchange derived from not irrigating certain lands
Boyle option 3	Ground water area undefined.	1,400 to 3,800 acre feet	Farm fields where well is located.	Pay farmers to put in wells to replace surface water.
Riverside Drains	Shallow ground water below Dawson Gothenburg and Kearney canals	8,000 acre feet	Platte River or tributaries at points near the river.	Could be used directly to feed wet meadows in some cases.
Ground water recharge	Excess flows in the North Platte River through Dawson and Gothenburg canals	1,000 acre feet/mo May-Sept (about 3-4 ft^3/s)	Platte River	Tamarack-like scheme that would increase flooding of agricultural fields and basements.

Alternatively, there is only one well that was sampled in the site to the east. Neither of the samples exceeded the irrigation criterion or the drinking water standard. However, both exceeded 5 µg/L and the potential recharge area is surrounded by sample sites that had selenium concentrations that exceeded 5 µg/L (Figure 166).   Since the plan would not directly involve flows to the river, the selenium would not directly affect the river. However, there would be return flows from the use of the water for irrigation. The return flows would either enter surface water tributaries or be returned to the ground water through deep percolation. Depending on irrigation efficiency, the selenium could be concentrated by a factor of 2 to 4 times in the return

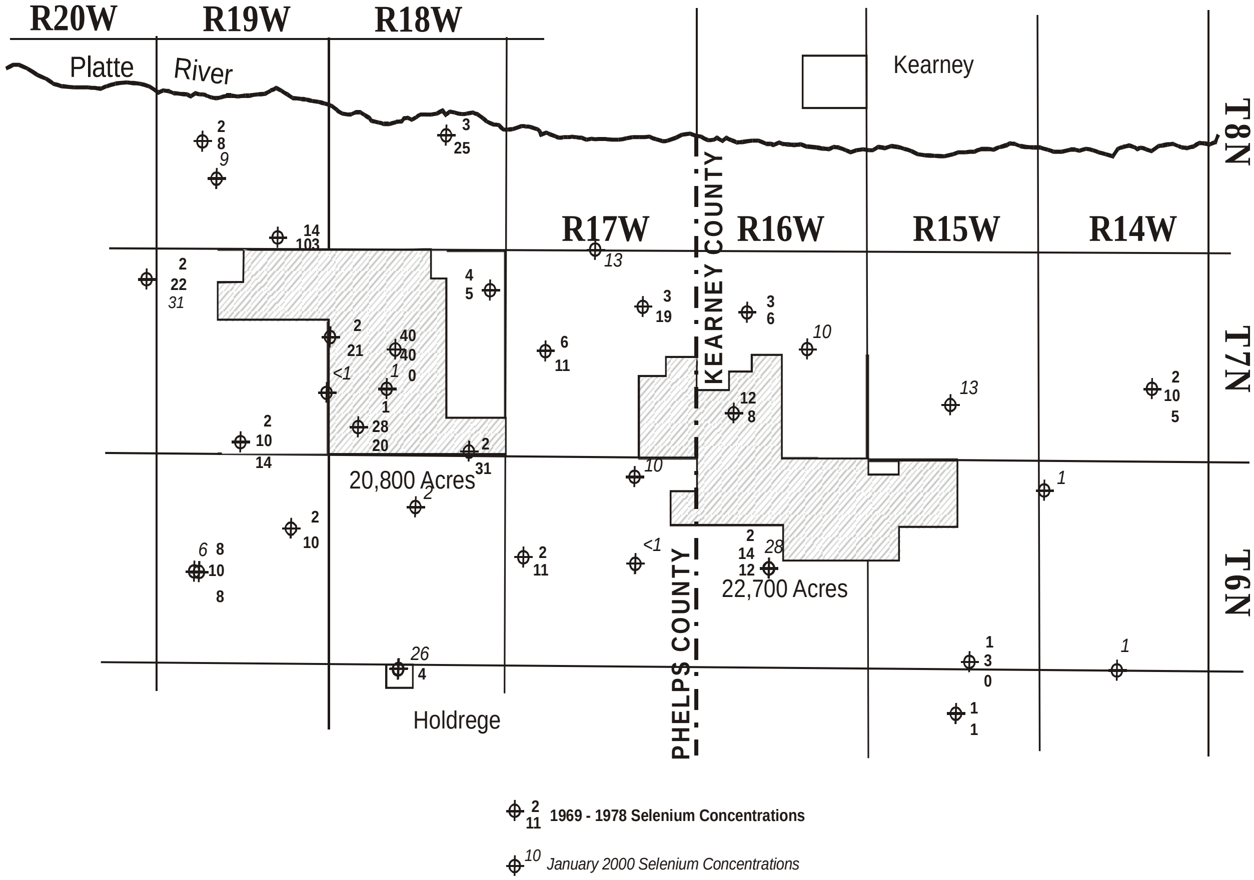


Figure 166. Ground water selenium concentrations in the vicinity of potential recharge sites in the Reclamation Ground Water Mound storage element

flows. If the selenium in the irrigation water is high, the return flows would have the potential to exceed the water quality standard for selenium. As noted earlier, most of the tributaries to the Platte River located near the ground water mound are classified for aquatic life support, for which the selenium standard is 5 µg/L.

There is also a ground water quality standard for selenium of 20 µg/L. The standard is exceeded in a number of the sampled wells, both during the late-1960's and early 1970's, as well in samples collected in January 2000. If the program were to degrade the water quality further. This could have implications related to compliance with water quality regulations and the Clean Water Act.

One other consideration in the use of recharge in the ground water mound is the relationship between NO_3–N and selenium. Presumably, the source of the ground water NO_3–N is in fertilizer, and the source of the selenium is in the soil. In the case of recharge, no fertilizer would be added. As was noted above, recent research has indicate that selenium can be mobilized by NO_3–N in fertilizer (Wright, 1999). Since there would be no fertilizer added to the recharge water, it may be that the recharge would not leach selenium to the ground water. Consequently it may be that the selenium in the recharge area would decrease either by mixing with or displacement by the low (or lower) selenium recharge water. This could cause a reduction in the source ground water over time. This is especially true if the recharge area is separated from the irrigated area.

The USBR plan would require more site-specific ground water data on selenium before it could be implemented. It would likely also require considerable coordination with agencies responsible for compliance with the Clean Water Act. Continued monitoring to assure that the selenium remained at acceptable concentrations would also seem to be a probable requirement if the plan were to be implemented.

The three Boyle (Boyle Engineering) options are taken from the States' Water Action Plan. Each proposal involves ground water management in one form or another. The areas include either one near Holdrege or another area to the northwest in Gosper County. The area near Holdrege is within the high selenium area defined by both sets of samples shown in figures 158 and 159. The data shown in Figure 158 indicate that the Gosper County site is within the low selenium (< 5 μg/L) part of the ground water mound.

The first Boyle option (Table 61) would involve pumping ground water from the first of the management areas described above and running it down North Dry Creek, either directly or down its tributary, Lost Creek. Selenium in the ground water in this area of the ground water mound ranges from 1 μg/L to as high as 26 μg/L. It would be generally south of the area included in the USBR plan. North Dry Creek is classified for aquatic life support. As was noted above, the selenium standard has been exceeded in the only sample collected, but the predicted data from the NO_3-N/selenium regressions indicates that the standard is probably exceeded frequently. Adding additional high selenium water to the creek would probably not affect this condition greatly. However, if the creek is not in compliance with the water quality standard, other complications could arise related to Clean Water Act compliance. This proposal would probably face requirements similar to, or more stringent than, those of the USBR plan, although there is no direct discharge to surface water under the USBR plan.

The second Boyle option would involve paying farmers to dry land farm for a year. The saved water would be held in the CNPPID system and released to the river for flow augmentation. From a water quality perspective, this option would not have any problems associated with the source water and could have a small favorable effect on water quality by slightly reducing one of the water uses most responsible for water quality degradation in the Middle Platte Basin, *i.e.* irrigation, as defined in NDEQ (1996c).

The third option is very similar to the USBR plan. The source area for the ground water is undefined, but since there are few areas in the mound that have low selenium, the probability seems high that water with excessive selenium would be encountered. With this in mind, this option would have the same problems and effects as the USBR plan, but the affected area would probability be somewhat different. Based on the introduction to the options, the affected area would be somewhat farther south, but still within the North Dry Creek basin.

The final two plans in Table 61 would each use distribution facilities to the north of the Platte River. In other words they are not directly involved with the ground water mound. The Riverside Drains would drain shallow ground water either directly to the river to augment flow or to wet meadows to improve riparian habitat and subsequently to the river or the alluvial ground water. The recharge plan would add ground water to the same general area as the drains for subsequent use by the program in somewhat similar fashion to Colorado's proposed Tamarack plan.

Platte River North Ground Water Quality

The preceding sections on ground water have dealt only with the ground water mound. Data on the area to the north of the river has been presented. As an introduction to the evaluation of the potential water quality implications of the plans for ground water management north of the Platte River, the water quality there will be briefly characterized. The water quality data for ground water on the north side of the river are summarized in Table 62, which also shows ground water standards and a comparison of the data to those standards.

Only a few of the water quality standards are exceeded by the ground water on the north side of the river (Table 62). The TDS standard was exceeded most often (58%), followed by NO_3–N (20%) and specific conductance (10%). Aside from atrazine, which exceeded the standard in 5 samples (5%), pH was greater than its standard once and below the lower end of the standard range in one sample. Aluminum and copper exceeded their respective standards once each. Based on the above, it would appear that the water quality on the north side of the river is somewhat better than that to the south in the mound, at least in comparison to water quality standards.

The selenium standard for aquatic life, which may come into play of the water is discharge to the river or wet meadow complexes, was not exceeded in wells on the north side of the river. However, there were only 9 samples that were analyzed for selenium. These constitute 3 samples from each of three wells. The samples were collected in 1970, 1973, and 1978. The maximum selenium was right at the standard of 5 µg/L. Although there are no problems with toxicity to aquatic life indicated, the number of samples is small and the areal coverage is extremely limited. Consequently, it would probably be a good idea to collect some additional samples in the vicinity of either of the plans before implementing ground water management to the north of the river.

Table 62. Summary of water quality data and comparison to standards for wells in Dawson and Buffalo counties

Statistic	Years	Water Temp. [°C]	Specific Conductance [µmho/cm]	D.O. [mg/L]	pH SU	TDS [mg/L]
Minimum	1936	9	260	0	6.47	195
Median	—	13	1100	4.8	7.3	637
Maximum	1993	20	3170	11.1	8.6	1970
No. of Obs.	642	451	534	90	368	199
No. < D.L.	—	—	—	—	—	N/A
Standard	—		2000	None	6.5-8.5	500
# > Std.	—	—	24	—	2	115

	NO_3-N diss. [mg/L]	Aluminum Al, diss. [µg/L]	Arsenic As, diss. [µg/L]	Barium Ba, diss. [µg/L]	Boron B, diss. [µg/L]	Chromium Cr, diss. [µg/L]
Minimum	0	0	0	0	0	0
Median	2.2	100	2	100	90	0
Maximum	83	400	16	400	590	20
No. of Obs.	414	19	19	19	198	16
No. < D.L.	0	15	1	12	1	16
Standard	10	50	50	1000	700	100
# > Std.	83	1	0	0	0	0

	Cobalt Co, diss. [µg/L]	Copper Cu, diss. [µg/L]	Lead Pb, diss. [µg/L]	Lithium Li, diss. [µg/L]	Nickel Ni, diss. [µg/L]	Selenium Se, diss. [µg/L]
Minimum	0	0	0	13	0	0
Median	0	3	2	25	0	1
Maximum	2	70	19	60	3	5
No. of Obs.	19	45	19	19	19	9
No. < D.L.	16	10	6	0	9	3
Standard	NA	67	50	None	19	5
# > Std.	NA	1	0	—	0	0

	Silver Ag, diss. [µg/L]	Vanadium V, diss. [µg/L]	Zinc Zn, diss. [µg/L]	Atrazine [µg/L]		
Minimum	0	0	0	0		
Median	0	5	30	0		
Maximum	0	14	1700	6.2		
No. of Obs.	9	19	43	108		
No. < D.L.	0	0	12	10		
Standard	50	None	2544	3		
# > Std.	0	—	0	5		

Comparison of ground water north and south of the Central Platte River

Table 63 presents a summary of contemporary ground water data from wells located in Dawson, Buffalo, and Hall counties on the north side of the Platte River and from Gosper, Phelps, and Kearney counties on the south side of the river. Table 63 includes data on water temperature, EC, three measures of nitrate, and four triazine herbicides.

The EC of the ground water on the north side of the river is nearly twice that of the water to the south (table 63). The ground water to the north is also higher in EC than the Overton/Grand Island river reach water, while the EC of ground water to the south is lower than that of the river (compare tables 44 and 63). There is no drinking water standard for EC, but the drinking water standard for TDS is 500 mg/L, which is approximately an EC of 750 µmho/cm. The median EC of the ground water to the north of the river is greater than the drinking water equivalent, while that of the ground water to the south of the river is lower (table 63).

The median temperature of the ground water to the north and south of the river (table 63) is about the same (13 and 14EC). The median ground water temperatures are approximately the same as the median temperatures of the river (compare to table 33). There is, however, a large difference in the minimum and maximum temperatures of the surface water and the ground water. The ground water has a much higher minimum temperature and a much lower maximum temperature than the surface water (compare tables 44 and 63). The maximum temperature of the ground water is well below the surface water temperature standard (table 63).

There are three measures of nitrate (NO_3) shown in table 63. Two of those include nitrite (NO_2), which is usually a transitory form that is in low to immeasurable concentrations. For this reason the $NO_2 + NO_3$ measures can usually be considered equivalent to NO_3. The inclusion of the $NO_2 + NO_3$ data expands the data base for NO_3 considerably; EPA gives a drinking water standard for $NO_2 + NO_3$, which is the same as that for nitrate alone.

There is measurable nitrate (*i.e.*, > the detection limit or D.L.) in virtually all of the ground water in the Middle Platte Valley (table II-2). The data indicate that around 20 percent to as many as 35 percent (total $NO_2 + NO_3$) of the ground water samples on the north side of the river exceed the drinking water standard. This amount is similar to the exceedence percentage that Gosselin *et al.* (1996) reported in domestic wells in the Platte Valley. The percentage of samples that exceed the nitrate standard on the south side of the river is much lower (table 63). Another great difference in the nitrate concentrations to the north and south of the river are in the maxima. Maximum nitrates to the north are greater than 80 mg/L for the total concentrations, but less than 50 and 20 mg/L for the total NO_3 and $NO_2 + NO_3$ respectively to the south. Excessive nitrate in ground water appears to be a problem throughout the valley, but more so to the north of the river.

Aquatic life criteria are applicable to surface water, but the most restrictive use of ground water, from a water quality perspective, is for domestic purposes. The aquatic life criterion for atrazine was given above as 1 µg/L; the drinking water standard is 3 µg/L. Drinking water standards for

Table 63.—Summary of Selected Ground Water Quality Data						
Area		Water Temp. (°C)	E.C. at 25°C (µmho/cm)	NO_3-N TOTAL (mg/L)	NO_2 & NO_3 N-total (mg/L)	NO_2 & NO_3 N-diss (mg/L)
North	Minimum	8.0	260	< 0.05	< 0.05	0.01
	Median	13.0	1,140	5.70	4.50	2.85
	Maximum	20.5	2,820	83.0	84.0	71.0
	No. of Obs.	1,183	220	382	1,337	292
	No. > D.L.	-----	-----	382	1,331	272
	No. > Std.	-----	-----	85	462	54
South	Minimum	10.0	170	< 0.1	0.01	< 0.01
	Median	14.0	657	2.7	4.25	2.90
	Maximum	24.0	1,970	47.0	17.0	14.0
	No. of Obs.	180	258	61	44	83
	No. > D.L.	-----	-----	61	43	83
	No. > Std.	-----	-----	3	3	4
Area		Atrazine Whole Sample (µg/L)	Atrazine Diss. (ppb)	Cyanazine Whole Water (µg/L)	Propazine Diss. (µg/L)	Simazine Whole Water (µg/L)
North	Minimum	< 0.05	< 0.05	0.02	0.05	0.01
	Median	0.70	0.05	0.10	0.05	0.05
	Maximum	8.55	0.19	0.84	0.05	0.70
	No. of Obs.	885	9	83	9	882
	No. > D.L.	874	4	1	0	101
	No. > Std.	23	0	0	0	0
South	Minimum	0.04	-----	0.10	-----	0.05
	Median	0.10	-----	0.40	-----	0.05
	Maximum	1.0	-----	0.4	-----	0.1
	No. of Obs.	42	0	42	0	42
	No. > D.L.	17	-----	0	-----	2
	No. > Std.	0	-----	0	-----	0
	Standard	3	3	1	20 or 200	4

the other triazine herbicides shown in table 63 are also shown at the bottom of the table. The only one of the herbicides that exceeds its drinking water standard in the Platte Valley is atrazine, which is present in measurable concentrations in nearly 100 percent of the wells sampled to the north of the river (table 63). About 3 percent of the samples exceeded the drinking water standard.

Gosselin et al., (1996) compared nitrate data from 1984–89 and 1994–95. They found no significant difference between either nitrates or atrazine between the two periods. They did find a positive statistical association between nitrate contamination and atrazine contamination. They also found an association between the depth of the well and the degree of contamination; shallower wells were more likely to be contaminated than deeper wells.

Sediment related comparison of alternatives

Affected Environment

The USGS collected sediment samples for chemical analysis in the Platte River at the Overton and Grand Island gages during September 1993. These data have been retrieved from EPA's STORET database. There are no sediment quality standards comparable to water quality standards at present. Consequently another basis for evaluating sediments for potential contamination is necessary. Table 63 presents various baseline concentrations for elements in soils and rocks. The Department of the Interior's National Irrigation Water Quality Program (NIWQP) uses the Western Soils baseline (Shacklette and Boerngen, 1984) to evaluate potential sediment contamination on the assumption that stream sediment is derived from soil erosion and that such a database would provide a reasonable basis for comparison (Severson, et al., 1991). The soils baseline is based on the geometric mean " 2 geometric deviations of samples collected throughout the 17 Western States.

Table 63 also shows baseline concentrations in various rocks. The first is based on crustal abundance values (CAV) from Fortescue (1992). CAV's are used in exploration geochemistry to identify potential ore-bearing sediments; enrichment is defined by concentrations 3 or more times the CAV (Church *et al.*, 1997). Enrichment could also define contamination. The CAV is based on average concentrations of the elements in various rocks assuming the composition of the earth's crust to be that of Clarke, i.e., 95 percent igneous, 4 percent shale, 0.75 percent sandstone, and 0.25 percent limestone (Parker, 1967). The average concentrations of the elements in various sedimentary rocks from Parker (1967) are also included in table 63 to illustrate the variability of their concentration in different natural sources.

The concentrations of 41 elements in 3 Platte River sediment samples are shown in table 64. In general, the elements show a continual increase or decrease in the downstream direction or there is no difference in the concentration at the 3 sites. However, several of the elements, including chromium, iron, mercury, selenium, uranium, vanadium, and zirconium, show no directional trend and have their maximum at Overton. Alternatively, thorium shows a minimum at Overton.

A large number of the elements in shown in Table 64 exceed the geometric mean concentration in the baseline for soils, indicating that the concentration are above average relative to the Western U.S. However, this result does not mean that the elements are elevated in the sediments, only that they are above average. A better indicator of elevated concentrations would be the 95 percent confidence interval for the baseline. A comparison of the two Platte River samples to the upper 95 percent confidence limit of the elemental data base for Western Soils shows that 6 are elevated at the Brady gage and 5 are elevated at the Overton and Grand Island gages. Those elements that are elevated relative to the baseline at all sample sites include manganese, niobium, phosphorus, selenium, and uranium (Table 64). Thorium is elevated at the Brady site only (compare tables 67 and 68).

Table 63: Various baselines for comparison to Platte River sediment data

Element	Units	[CAV] Clarke {ppm}	Concentration in various rocks (ppm)				Soils Baseline		
			Clays and shales	Shales	Sandstones	Carbonate rocks	Geometric Mean	Western States	
								Lower C.I.	Upper C.I.
Antimony	Sb ppm-s	0.2	2	1.5	< 1	< 1	—	—	—
Arsenic	As ppm-s	1.8	6.6	13	1	1	5.5	1.2	22
Beryllium	Be ppm-s	2	3	3	< 1	< 1	0.68	0.13	3.6
Bismuth	Bi ppm-s	0.0082	0.01	—	—	—	—	—	—
Boron	B ppm-s	9	100	100	35	20	23	5.8	91
Cadmium	Cd ppm-s	0.16	0.3	0.3	< 0.1	0.035	—	—	—
Cerium	Ce ppm-s	66.4	50	59	92	11.5	65	22	190
Chromium	Cr ppm-s	122	100	90	35	11	41	8.5	200
Cobalt	Co ppm-s	29	20	19	0.3	0.1	7.1	1.8	28
Copper	Cu ppm-s	68	57	45	< 1	4	21	4.9	90
Europium.	Eu ppm-s	2.14	1	1	1.6	0.2	—	—	—
Gallium	Ga ppm-s	19	30	19	12	4	16	5.7	45
Gold	Au ppm-s	0.004	0.001	< 0.01	< 0.01	< 0.01	—	—	—
Holmium	Ho ppm-s	1.26	1	1.2	2	0.3	—	—	—
Iron	Fe %-s	6.22	3.33	4.72	0.98	0.38	—	—	—
Lanthanum	La ppm-s	34.6	40	92	30	< 1	—	—	—
Lead	Pb ppm-s	13	20	20	7	9	17	5.2	55
Lithium	Li ppm-s	18	60	66	15	5	—	—	—
Magnesium	Mg %-s	2.76	1.34	1.5	0.7	4.7	0.74	0.15	3.6
Manganese	Mn ppm-s	1060	670	850	< 100	1100	380	97	1500
Mercury	Hg,ppm	0.086	0.4	0.4	0.03	0.04	0.046	0.0085	0.25
Molybdenum	Mo ppm-s	1.2	2	2.6	0.2	0.4	0.85	0.18	4
Neodymium.	Nd ppm-s	39.6	23	24	37	4.7	36	12	110
Nickel	Ni ppm-s	99	95	68	0.2	20	15	3.4	66
Niobium.	Nb ppm-s	20	20	11	< 0.1	0.3	—	—	—
Phosphorus	P %-s	0.112	0.077	0.07	0.017	0.04	0.032	0.0059	0.17
Potassium	K %-s	1.84	2.28	2.66	1.07	0.27	1.8	0.38	3.2
Scandium	Sc ppm-s	25	10	13	1	1	8.2	2.7	25
Selenium	Se ppm	0.05	0.6	0.6	0.05	0.08	0.23	0.039	1.4
Silver	Ag ppm-s	0.08	0.1	0.07	< 0.1	< 0.1	0.5	—	—
Sodium	Na %-s	2.27	0.66	0.96	0.33	0.04	—	—	—
Strontium	Sr ppm-s	384	450	300	20	610	200	43	930
Sulfur	S	340	3000	2400	240	1200	—	—	—
Tantalum	Ta ppm-s	1.7	3.5	0.8	< 0.1	< 0.1	—	—	—
Thorium.	Th ppm-s	8.1	11	12	1.7	1.7	9.1	4.1	20
Tin	Sn ppm-s	2.1	10	6	< 1	< 1	—	—	—
Uranium	U ppm-s	2.3	3.2	3.7	0.45	2.2	2.5	1.2	5.3
Vanadium	V ppm-s	136	130	130	20	20	70	18	270
Yttrium	Y ppm-s	31	30	26	40	30	22	8	60
Zinc	Zn ppm-s	76	80	95	16	20	55	17	180
Zirconium .	Zr ppm-s	162	200	160	220	19	—	—	—

Table 64. USGS Data from NAWQA Study - All Dry Weight

Element	Units	Brady	Overton	Grand Island	Element	Units	Brady	Overton	Grand Island
Antimony	Sb ppm	0.9	0.9	0.9	Molybdenum	Mo ppm	< 3.5	< 3.5	< 3.5
Arsenic	As ppm	7.2	9.1	9.1	Neodymium.	Nd ppm	67	49	47
Beryllium	Be ppm	1.8	1.8	1.8	Nickel	Ni ppm	23	35	35
Bismuth	Bi ppm	< 18	< 18	< 18	Niobium.	Nb ppm	14	14	14
Boron	B ppm	11	16	19	Phosphorus	P %	0.2	0.2	0.3
Cadmium	Cd ppm	0.7	1.1	1.1	Potassium	K %	2.8	2.5	2.3
Cerium	Ce ppm	149	104	100	Scandium	Sc ppm	11	11	11
Chromium	Cr ppm	60	74	65	Selenium	Se ppm	2.1	4.6	3.7
Cobalt	Co ppm	14	19	21	Silver	Ag ppm	0.2	0.4	0.7
Copper	Cu ppm	25	35	39	Sodium	Na %	1.6	0.9	0.8
Europium.	Eu ppm	81	56	53	Strontium	Sr ppm	667	737	877
Gallium	Ga ppm	21	23	23	Sulfur	S ppm	0.3	0.7	0.6
Gold	Au ppm	< 14	< 14	< 14	Tantalum	Ta ppm	< 70	< 70	< 70
Holmium	Ho ppm	<7	<7	<7	Thorium.	Th ppm	25	18	19
Iron	Fe %	3.2	3.7	3.5	Tin	Sn ppm	< 18	< 18	< 18
Lanthanum	La ppm	81	56	53	Uranium	U ppm	10	14	11
Lead	Pb ppm	33	40	46	Vanadium	V ppm	93	107	102
Lithium	Li ppm	35	35	35	Yttrium	Y ppm	37	30	28
Magnesium	Mg %	1.3	1.5	1.6	Zinc	Zn ppm	3.5	3.5	3.5
Manganese	Mn ppm	1,754	2,632	2,807	Zirconium .	Zr ppm	96	144	142
Mercury	Hg ppm	< 0.04	0.05	< 0.04					

Frenzel *et al.* (1998) evaluated the sediment data shown in Table 64 in the NAWQA study report. They indicated that streambed sediment samples had notable selenium concentrations. Frenzel *et al.* (1998) did not indicate that there were any other trace elements of concern in the Platte River sediments. Frenzel *et al.* (1998) also reported on additional samples collected at Brady and Grand Island following high flows in 1995. Those samples had 2 and 1.8 ppm of selenium respectively.

Selenium exceeded the upper limit of the western soils baseline in both of the Platte River sediment samples (tables 67 and 68). The selenium concentration in the sediments also exceeded the factor of three rule-of-thumb enrichment indicator. The NIWQP developed reference concentrations for selenium in sediments (NIWQP, 1998). The background concentration for selenium in selenium-normal environments was 0.1-2.0 ppm, which is the same as that of Pais and Jones (1997) for soils. However, selenium concentrations < 1 ppm have been associated with reproductive effects in birds nesting in shallow terminal ponds (NIWQP, 1998). Alternatively an EC_{10}[1] of 2.5 ppm of selenium was estimated for fish and birds in a variety of freshwater habitats (NIWQP, 1998).

The existence of a seleniferous area in the North Platte Basin has been known for a long time (PHS, 1951; Rosenfield and Beath, 1964; Crist, 1974). The upper basin is a possible source of the seleniferous sediments. However, there are two large reservoirs (Glendo Reservoir and Lake McConaughy) between that source area and the study area. The reservoirs should trap all, or nearly all, of the sediment that originates from the North Platte River in Wyoming.

There are outcrops of the Pierre Shale in the South Platte Basin in Colorado (Anderson *et al.*, 1961). In the Central Platte Basin, the Pierre Shale forms bedrock well below the ground surface (Peckenpaugh *et al.*, 1987; Peckenpaugh and Dugan, 1983) and on this basis would be an unlikely source of seleniferous sediments. Pierre Shale is known to be locally seleniferous (Rosenfield and Beath, 1964), although samples from the South Platte River showed only a maximum of 2 ppm of selenium (Anderson et al. (1961). Frenzel *et al.* (1998) noted that there were higher concentrations of selenium in upstream sediments in the Central Platte River than in those farther downstream. Table 64 shows a somewhat higher concentration of selenium in the Overton sample than in the one from Grand Island. This would indicate that the source is either upstream in the Platte or the South Platte or possibly both. However, because the geologic formations from which the selenium is most likely derived are at depth in the Platte Basin, the more likely source is the South Platte. The reduction at the downstream (Grand Island) site would be due to dilution by uncontaminated sediments from tributary inflows, similar to that which occurs in water. No matter what the source, there may be a concern for sediment selenium in the type of environment where selenium effects are favored, e.g. terminal wetlands. The only wetlands in the Central Platte Basin that would be possibly analogous to terminal

[1] The EC_{10} is the effective concentration at which 10 percent of the exposed population exhibit a particular effect or response. In the case of selenium the effect would be on avian reproduction (birth defects) and sublethal effects to adult fish along with reproductive effects.

wetlands would be those that are only filled seasonally by overbank flows. However, in terminal wetlands, the majority of the water loss is through evaporation, which concentrates dissolved substances, while the majority of the outflow from the alluvial wetlands described above is through seepage to the river, which carries dissolved substances to the river with it. The evaporative concentration in terminal wetlands is a major contributor to the adverse biological effects of contaminants in those environments.

The one other element in table 64 that exceeded the upper confidence limit of the Western soils baseline is uranium. Elevated uranium is often associated with elevated selenium, so much so that selenium indicator plants have been used in uranium prospecting (Cannon, 1960). Based on this association. the source of uranium is likely to be the same as the source of selenium. Uranium is also higher at Overton than at Grand Island indicating an upstream origin. High uranium concentrations have been reported in the alluvium in Morgan, Logan, and Boulder counties in eastern Colorado (CDPHE, 1999). All three counties are in the South Platte Basin.

The uranium concentration at Grand Island is 6 ppm, which is less than 3 times the CAV. On this basis, those sediments would not be considered enriched relative to the CAV. Alternatively, Pais and Jones (1997) give a baseline for soils that ranges from 0.1 to 11.2 ppm; on this basis, neither of the sediment samples would be considered elevated in uranium.

Uranium has not been found to be particularly toxic. For example, there is no EPA aquatic life criterion for uranium. Recent work on the reproductive effects of uranium in mice found effects (reduced litter size and increase number of stillbirths) only at the very highest doses (25-80 mg/Kg [ppm] per day of body weight: Corbella and Domingo, 1996). Similar results are reported for dogs and rats in EPA's IRIS (Integrated Risk Information System) database (EPA, 1989). Sediment uranium does not seem elevated enough to warrant particular concern in the Central Platte Basin.

Selenium is found in relatively high concentrations in fish and eggs of endangered birds in the Central Platte (see the section on contaminants in fish and bird eggs elsewhere in this report). Because of the concern over selenium and the small number of samples that had been collected in the Central Platte, the Program collected additional sediment samples for the analysis of trace elements. Sediment samples were collected for analysis of contaminants at numerous sites in the Platte River from the confluence of the North and South Platte rivers to Grand Island, along with 3 sites in the South Platte and 2 sites in the North Platte rivers. The sample site locations in the Central Platte Basin are shown on Figure 55. A summary of the data is presented in Table 65. The summary indicates that most of the sediment concentrations are below levels of concern, as indicated by screening levels (UET) shown in the last column of the table.

Table 65 shows the number of samples that exceed the detection limit (D.L.). The detection limit is the lowest concentration of an element that can be quantified by the analytical method used on the samples. The D.L. is represented by the minimum concentration shown for the elements that have a result less than the D.L. In some cases the D.L. varies, as is illustrated by the mercury results, where the D.L. ranged from 0.019 to 0.040. The variation in the D.L. can relate to either a difference in the size of samples used in the pre-analysis extraction or matrix interference to the instrument reading.

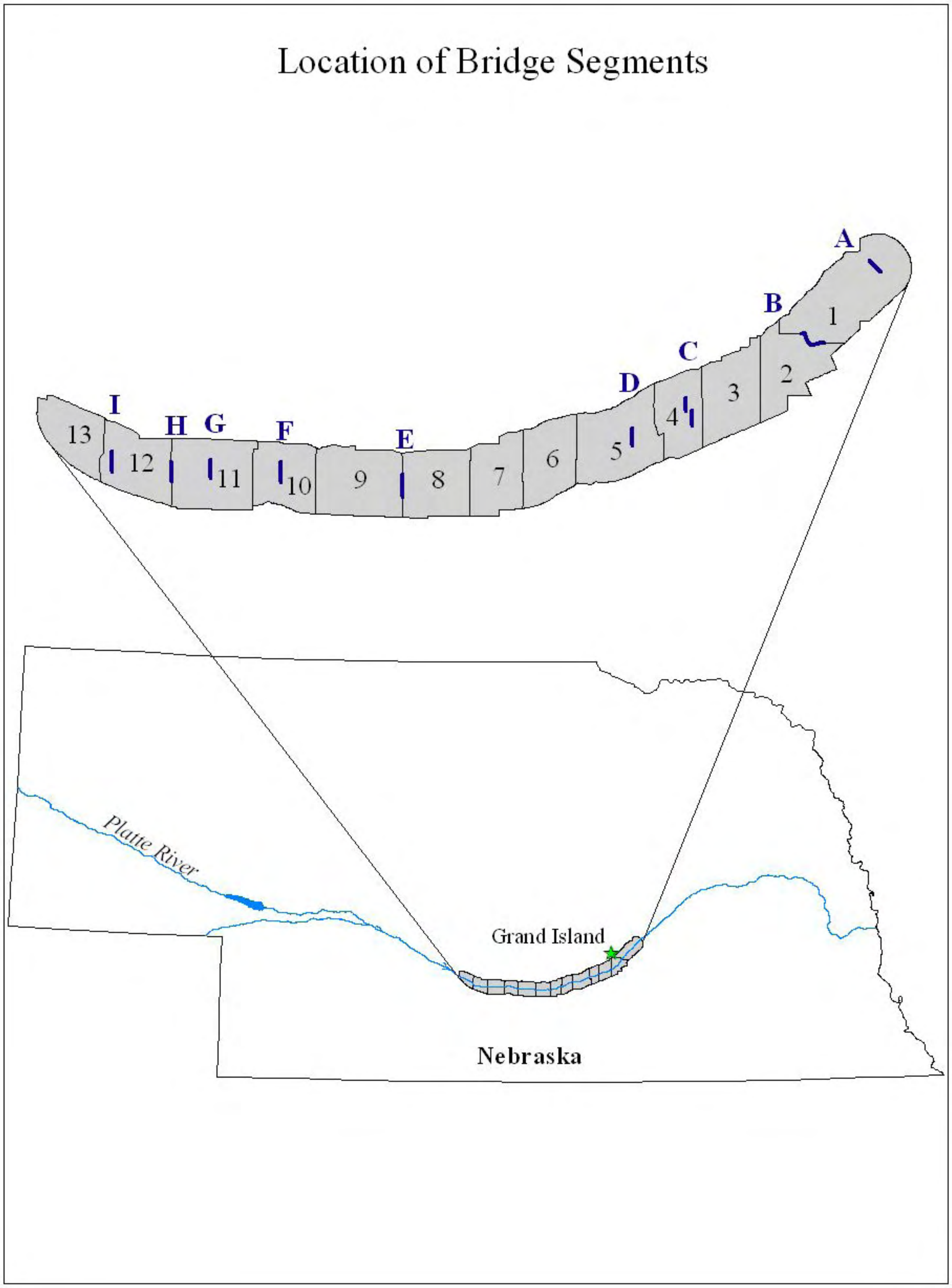


Figure 55. Location of Middle Platte Basin sediment chemistry sample sites

Table 65. Summary of Platte River sediment chemical analysis (ppm) for samples collected in January 2000							
Element	Symbol	No. of Obs.	No. > D.L.	Minimum	Median	Maximum	UET[1]
Silver	Ag	43	2	< 0.8	< 0.8	1.8	4.5
Aluminum	Al	43	43	434	2,515	13,036	—
Arsenic	As	27	27	0.4	0.8	3.1	17
Barium	Ba	43	43	8	46	148	—
Beryllium	Be	43	36	< 0.1	0.2	0.9	—
Calcium	Ca	43	43	780	4,673	23,228	—
Cadmium	Cd	43	1	< 0.8	< 0.8	2.4	3.0
Cobalt	Co	43	41	< 0.6	3.1	10.5	—
Chromium	Cr	43	43	1.0	6.2	30.6	95
Copper	Cu	43	43	60	766	23,006	86
Mercury	Hg	27	0	< 0.019	< 0.020	< 0.040	0.56
Iron	Fe	43	43	1,227	5,589	28,037	40,000
Potassium	K	43	42	< 200	767	3,115	—
Magnesium	Mg	43	43	231	1,090	4,596	—
Manganese	Mn	43	43	24	152	815	1,100
Molybdenum	Mo	43	1	< 2.0	6.9	6.9	—
Sodium	Na	43	43	25	151	469	—
Nickel	Ni	43	41	< 2	6	146	43
Lead	Pb	43	41	< 6	15	295	127
Antimony	Sb	43	0	< 4	---	< 4	3.0
Selenium	Se	27	5	< 0.4	< 0.4	0.7	2.5[2]
Silicon	Si	43	38	0	602	4,626	—
Strontium	Sr	43	43	4	24	107	—
Vanadium	V	43	43	1	11	69	—
Zinc	Zn	43	43	19	115	4,000	520

[1] Upper Effects Threshold (NOAA, 1999)

[2] No UET. Concentration shown is that at which 10 percent of the fish and birds in a variety of aquatic systems show effects (NIWQP, 1998)

The main reason for collecting the additional sediment samples in 2000 was based on a concern over selenium; however, high concentrations of mercury were also observed in biological tissue samples from the Central Platte. None of the mercury samples in sediments collected in January 2000 were above the highest of its detection limits of approximately 0.04 ppm (Table 65) or less than 10 percent of the UET. In the case of selenium, the majority of the samples were below the D.L. (22 of 27), as is shown by the fact that the median is included in that group. Only 5 samples showed detectable selenium at a D.L. of 0.4 ppm or about 20 percent of the UET. Based on these results, mercury and selenium are not a concern due to sediment manipulation as part of any of the Platte River alternatives.

There are 4 of the elements in Table 65 that exceed their respective UET values and must be considered possible contaminants. These possible contaminants include copper, nickel, lead, and zinc. Copper shows the greatest exceedence in that the median concentration in the sediments exceeds the UET by a factor of 9. In the case of the other possible contaminants, the median is less than the UET, but the maximum concentration exceeds the UET, indicating further evaluation should be undertaken.

Environmental Consequences

Method of analysis

The effects on concentrations of the various elements from the sediment chemistry data will be evaluated by weighting the chemical concentrations in proportion to the deposition of sediment at augmentation sites as defined by the SEDVEG model. The output from the sediment-vegetation model includes data on 10 different size fractions of sediment (Table 66). In each particle-size range, the upper limit of the range is double its lower limit. The sediment samples collected for chemical analysis were in the less than 2 mm size range. To be consistent with the chemical data, only the first 7 size fractions shown in Table 66 are being used in the contaminants assessment.

Table 66. Sediment-vegetation model size fractions

Fraction	Particle size (mm)
1	0.016-0.031
2	0.031-0.062
3	0.062-0.125
4	0.125-0.250
5	0.25-0.50
6	0.50-1.0
7	1.0-2.0
8	2.0-4.0
9	4.0-8.0
10	8.0-16.

The changes in the concentrations of the elements in the bed sediments will be estimated based on the assumption that added sediment will be similar in quality to that of local bank or island sediments where augmentation occurs. Augmentation occurs at cross-sections 3 and 19 in the SedVeg Model. Augmentation will also occur due to island leveling, in which case the added sediment will be of similar quality to the islands in cross-sections where the island leveling occurs. Island leveling occurs at Cross-sections 22 and 56 in the SedVeg Model. The quality of bed sediments in the cross-sections to be evaluated are shown in Table 67. The quality of added sediments (bank or island) are also shown in Table 67. The average of all of the depths from both the USGS and USBR columns in Table 67 was used as an estimate of the quality of the added sediment.

There are 2 headings under each of the metals shown in Table 67. The headings represent the laboratories that performed the analysis. Samples from all sites were analyzed at the Reclamation (USBR in Table 67) laboratory. Samples at selected sites were split and submitted to the U.S. Mineral Resource Surveys Program laboratory (USGS) laboratory for quality assurance. Although there appears to be some relatively large differences between the results from the different laboratories, these are generally within acceptable limits. Much of the variation is likely due to the samples themselves. Unlike water, where dissolved solids tend to be evenly dispersed, sediments are solids and chemical constituents are not evenly dispersed. Although every effort was made to process the samples such that an even distribution of chemical constituents occurred in the subsamples, a great deal of variation among them is expected.

Table 67. Platte River sediment concentrations for analysis of alternatives for toxicity										
Sample site - Cross-section	Source	Location/ Depth	Copper		Nickel		Lead		Zinc	
			USGS mg/kg	USBR ppm	USGS mg/kg	USBR ppm	USGS mg/kg	USBR ppm	USGS mg/kg	USBR ppm
Lexington - XS 3	In-Channel	North	365	389	3.81	4.05	13.21	11.0	72.6	90.7
	In-Channel	South	4129	4364	11.31	12.2	27.4	31.1	575	802
	Bank	0' Depth	—	462	—	5.48	—	9.50	—	101
		1' Depth	2032	2112	12.49	14.0	17.2	22.9	305	406
		2' Depth	—	766	—	6.54	—	29.0	—	167
Elm Creek Bridge. - XS 19	In-Channel	North	1388	1456	3.63	5.46	10.88	12.1	214	285
	In-Channel	South	961	978	3.61	4.08	9.61	13.1	152	195
	Bank	0' Depth	—	331	—	5.15	—	6.25	—	79.0
		1' Depth	297	306	4.44	5.42	5.99	7.27	58.3	70.7
		2' Depth	—	214	—	7.43	—	11.9	—	62.6
Kearney Bridge - XS 22	In-Channel	Main	780	780	2.13	2.00	9.4	13.3	113	150
	Bank	0' Depth	—	2035	—	13.12	—	32.2	—	405
		1' Depth	9014	10020	9.05	10.30	44.8	48.9	1680	2367
		2' Depth	—	7412	—	19.08	—	162.3	—	1632
	Island	0' Depth	—	1032	—	8.10	—	15.5	—	210
		1' Depth	806	828	3.79	4.94	8.3	13.1	123	162
		2' Depth	—	615	—	5.61	—	20.9	—	125
near Alda - XS 56	In-Channel	Main	412	485	3.09	3.49	13.0	12.6	73.3	57.6
	Island	0' Depth	—	2456	—	6.20	—	17.5	—	46.4
		1' Depth	4099	4554	8.55	9.17	18.6	21.2	646	46.7
		2' Depth	—	1238	—	4.28	—	18.3	—	42.2
	In-Channel	North	575	633	3.19	3.73	8.10	8.72	102	47.1
	Bank	0' Depth	—	433	—	9.34	—	15.0	—	96.2
		1' Depth	350	417	9.56	10.6	7.83	10.5	66.7	148
		2' Depth	—	245	—	8.13	—	9.56	—	115

The effect on the concentration of the above 4 elements in the sediments with any alternative will be adjusted when deposition occurs. The cumulative and annual deposition is generated by the SedVeg Model. In that data file, the deposition or erosion is included as a positive value when the concentration of sediment is positive and negative when erosion occurs. When erosion occurs, no adjustment to the sediment concentration is made. When deposition occurs, the sediment concentration is adjusted based on the amount of deposition in tons, based on the following formula:

$$C_{alt} = (V_s * C_{bed} + V_d * C_{bank}) / (V_s + V_d),$$

where C_{alt} = the concentration with the alternative,
V_s = the sediment load,
C_{bed} = the concentration in the bed sediments,
V_d = the mass of sediment deposited with the alternative,
C_{bank} = the concentration in the bank (or island) sediments.

The concentration of each element is reported at the beginning, after 13 years, and at the end of the total SedVeg Model simulation period. The initial concentration is equivalent to the Present Condition. Changes then reflect the effects of the alternative.

EPA (2004) has derived a series of logistic regression curves for sediment contaminants. There are curves for each of the above listed elements. The regressions estimate the probability of toxicity for a continuum of concentrations of each of the elements. These probabilities will be the basis for the impact assessment in the FEIS. The probabilities of toxicity will be calculated from the concentrations of copper, nickel, lead, and zinc at cross- sections 3, 19, 22, and 56 in the Platte River SedVeg output. For each alternative, the concentrations of each of the elements in that reach of the river as estimated above will be used to calculate the probability of toxicity from the logistic regression for that element. At times of no augmentation, the probability of toxicity (and the elemental concentration) would remain unchanged. The change in the probabilities of toxicity due to copper, nickel, lead, and zinc would be the basis for the impact assessment.

The final toxicity estimates should not be taken as any more than qualitative. All that the method can say is that if the local sediment used for augmentation has a higher concentration than the bed sediments, the concentration in the bed sediments is likely to increase, and if the local sediment used for augmentation has a lower concentration than the bed sediments, the concentration in the bed sediments is likely to decrease The changes in concentrations and the associated probabilities of toxicity are developed as a means of comparing alternatives only. The actual changes in the concentrations contaminants in the bed sediments and the associated probability of toxicity will be based on much more complex interactions than the above analytical method could simulate.

Present condition

The data in Table 65 lump all of the data from the river, bank, and island samples. The only potentially toxic contaminants in the table, based on UET screening, include copper, lead, nickel, and zinc. The data collected in 2000 will be used to represent the present condition. Emphasis will be on copper, which appears to be the best indicator, because copper exceeds its UET in the majority of samples collected in 2000. However, lead, nickel, and zinc will also be considered and carried through the analysis.

Table 166A shows a plot of the individual river bed copper sample results for each of the sites in the Platte River basin in Nebraska. Table 166 includes all sites, while Table 2 only includes sites important to evaluating the effects of sediment augmentation. The bed sediment sample sites coincide with the bridge segments for the endangered species habitat studies. At the scale of Table 166A, the UET barely sits above the abscissa, reflecting the degree to which the maximum copper concentration in the Platte River sediments exceeds the UET.

With the exception of the sample from the Sutherland (Korty) Canal diversion, the South Platte copper concentration exceeds that of the samples from the North Platte River (Table 166A). The increase in copper in the South Platte between the Sutherland diversion and the Highway 83 site should indicate that the source is between the 2 sites, although there is a degree of uncertainty in such an hypothesis. The concentration in the bed sediment samples shows where the sediment was deposited, not necessarily that it originated nearby. However, diversions such as Sutherland are usually preferred sites for sediment deposition, rather than erosion.

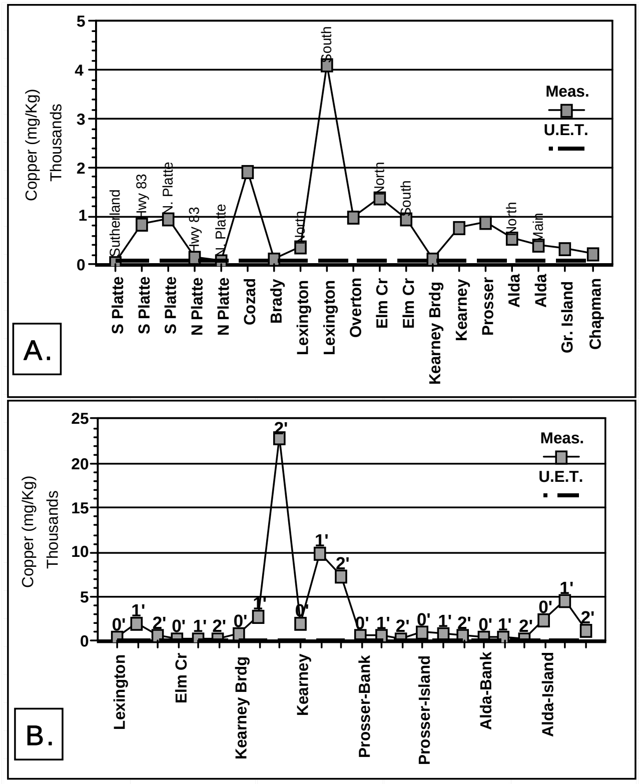


Table 166. Copper concentrations in bed (A.) and bank and island (B.) sediment samples from the Platte River basin in Nebraska

There is also a large increase in the copper concentration in the bed sediments at Cozad (Table 166A), which is the first site below the confluence of the North and South Platte rivers. The copper concentration at Brady, the next site on the Platte River, shows a large decrease from what was shown at Cozad. Samples were next collected from each of the north and south channels of the Platte River at Lexington. The samples from the 2 channels show the greatest difference among any pair of samples from adjacent sites in the basin (Table 166A). The copper concentration in the sample from the south channel at Lexington is the highest by far in the basin and exceeds the next highest concentration by a factor of 2. The copper concentration in the sample from Overton is intermediate between the samples from the channels at Lexington and much higher than the sample from the river near Brady (Table 166A). The samples collected from the north and south channels at the Elm Creek bridge are similar in copper concentrations, although the sample from the north channel is somewhat higher than the one from the south channel. The copper concentration drops to near the UET in the sample from the Kearney Bridge, followed by increases at Kearney and Prosser. The copper concentration then decreases between Prosser and Chapman. However, the copper concentration in the sample from Chapman is still greater than the UET.

Table 166B shows the copper concentrations in the bank and island samples from the Platte Basin. Three samples were collected at each of the bank/island sites. The samples were collected at 1−foot intervals from the surface to 3 feet deep, *i.e.* 0-1, 1-2, and 2-3 feet. The scale of the ordinate on Table 166B is 5 times that of Figure 24A. The *y*−axis scale is dictated by the copper concentration in the deepest bank sample collected at the Kearney Bridge, which had a copper concentration of over 23,000 mg/Kg (= ppm). The deeper samples collected from the river bank near Kearney also had concentrations in excess of 5,000 mg/Kg, which was the upper limit of the *y*-axis on Table 166A. The only other sample that approaches 5,000 mg/Kg is the 1−foot island sample from Alda, where the copper concentration was 4,554 mg/Kg. However, there are 6 other samples with copper concentrations between 1,000 and 4,000 ppm of copper.

This compares to 1 bed sample with a copper concentration greater than 4,000 ppm and 2 others with concentrations between 1,000 and 4,000 ppm.

Table 167A shows a plot of the nickel data from the bed sediments in the Platte River basin in Nebraska. As was the case with copper, the South Platte River sediments are higher in nickel than those from the North Platte River. With the exception of the south channel at Lexington, the sediment samples from the Platte River are also lower in nickel than the South Platte River sediments. The maximum nickel concentration in the bed sediments is only about ¼ of the UET. Nickel is apparently not currently a problem in the bed sediments in the Platte River basin under the existing conditions.

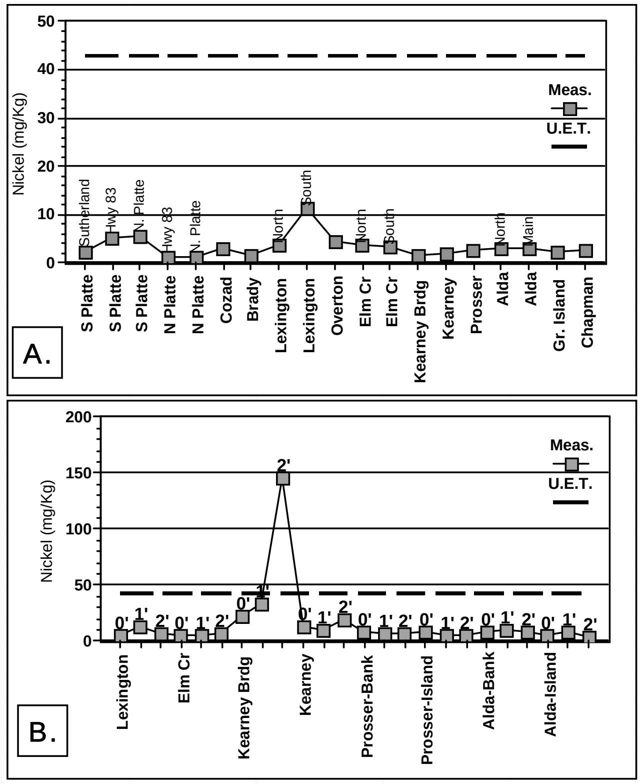


Table 167. Nickel concentrations in bed (A.) and bank and island (B.) sediment samples from the Platte River basin in Nebraska

Among the bank and island samples, only the deep sample from the river bank at the Kearney Bridge exceeds the nickel UET (Table 167B). This means that the only exceedence in Table 65 above was a reflection of the maximum concentration in the table. The only other bank or island sample that approached the UET was the 1–foot sample from the same site. The remaining bank and island samples are similar in nickel concentration to the bed samples and would similarly not pose a threat to aquatic life if the bank or island samples entered the river.

Lead concentrations in the bed sediments are shown on Table 168A. As is shown in the note on Table 168A, the UET is off the scale of the plot. As was the case with nickel in the bed sediments, the maximum lead concentration in the bed sediments is less than ¼ of the UET, and lead does not appear to pose a threat to benthic life in the bed sediments.

Lead is much higher in the South Platte River sediments than in those of the North Platte River (Table 168A). Alternatively, lead in the North Platte River sediments is lower than that in the Middle Plate River sediments. Both of the North Platte River sites are downstream from Lake McConaughy and would reflect local influences. Lead in the South Platte River sediments is also generally higher than in the Middle Platte River sediments. The lone exception is the 2nd site at Lexington (south channel), which has a lead concentration nearly as high as the maximum in the South Platte River sediments. The lower lead concentration in the Middle Platte River probably reflects the combined effects of removal at diversions and dilution by other sources of sediment downstream from the North Platte-South Platte confluence.

Two of the bank samples exceed the UET for lead (Table 168B). The samples are from the Kearney bridge and Kearney sites. In both cases, the samples that exceed the lead UET are deep samples (2-3 feet). All of the shallower samples are less than ½ of the UET for lead. Lead in the bank and island sediments does not appear to be a potential threat to aquatic life if it were to enter the Platte River.

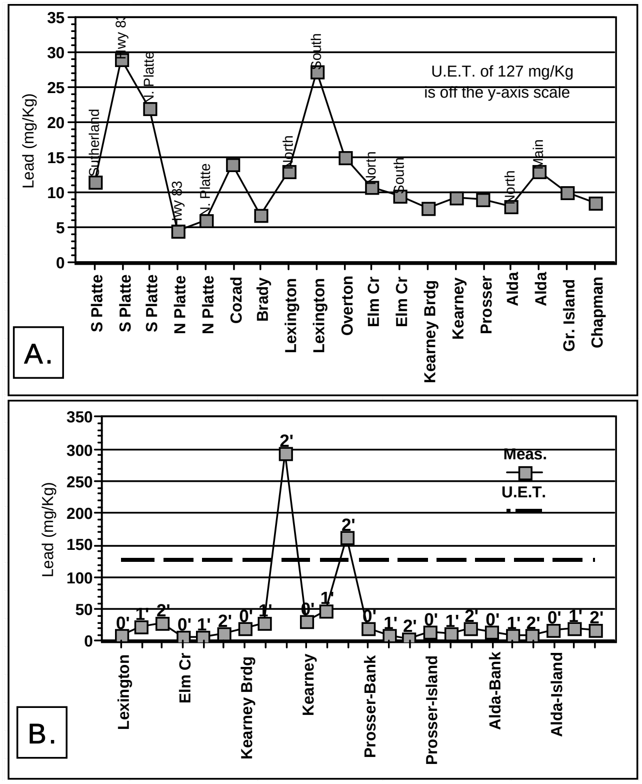


Table 168. Lead concentrations in bed (A.) and bank and island (B.) sediment samples from the Platte River basin in Nebraska

The only bed sediment sample that exceeds the zinc UET was collected from the south channel at the Lexington site (Table 169A). The south channel at the Lexington sample site had the maximum concentration of copper, nickel, and zinc of any of the bed sample collection sites, and was the 2^{nd} highest in lead after the South Platte River site at Highway 83 (see figures 56 through 59). For some unknown reason, the south channel of the Platte River near Lexington is the most contaminated site from the perspective of metals concentrations in the Middle Platte Basin.

Several bank samples from the Platte Basin exceeded the zinc UET (Table 169B). The highest zinc concentration in the bank samples came from the 2–foot or deep sample from the Kearney Bridge site. Both of the subsurface samples from the Kearney site also exceeded the zinc UET, with the 1–foot sample somewhat higher in zinc than the sample from the 2–foot depth. All of the remaining bank and island samples were well below the zinc UET, except for the 1–foot deep sample from the Lexington site, which at 406 mg/Kg was a little over 100 mg/Kg below the UET of 530 mg/Kg. Based on these results, the surface bank and island sediments are in the same range as the majority of the bed sediment samples as far as their zinc concentrations are concerned.

The metals data included on figures 56 through 59 seem to show considerable contagion among themselves. In other words, if one metal is high at a given site, all metals are high at the site. This is explored more fully in Table 68, which shows correlations among the various metals in the river bed samples and the combined bank and island samples. With the exception of the correlation between copper and zinc, those for the bank-island samples have higher r–values than those based on the river bed samples. The correlations for the bed samples, with the exception of the above-mentioned one for copper and zinc, have r–values around 0.7. Because the number of samples for the bed sediments is only 19, compared to 23 or 24 for the bank-island sediments, the significance levels for the bed samples are lower than those for the bank-island samples. The correlations based on the bank-island samples all have a significance level (probability of an r–value of the magnitude shown occurring by chance alone) of less than 1 in a million, compared to the bed samples that have significance levels between 2 in 1,000 and 2 in 10,000 (Table 68). The strong relationship among the various potential contaminants indicates that any increase or decrease in one metal should be accompanied by an increase or decrease in all of the 4 metals if the source is comparable to the banks or islands sampled in 2000 in the Platte Basin.

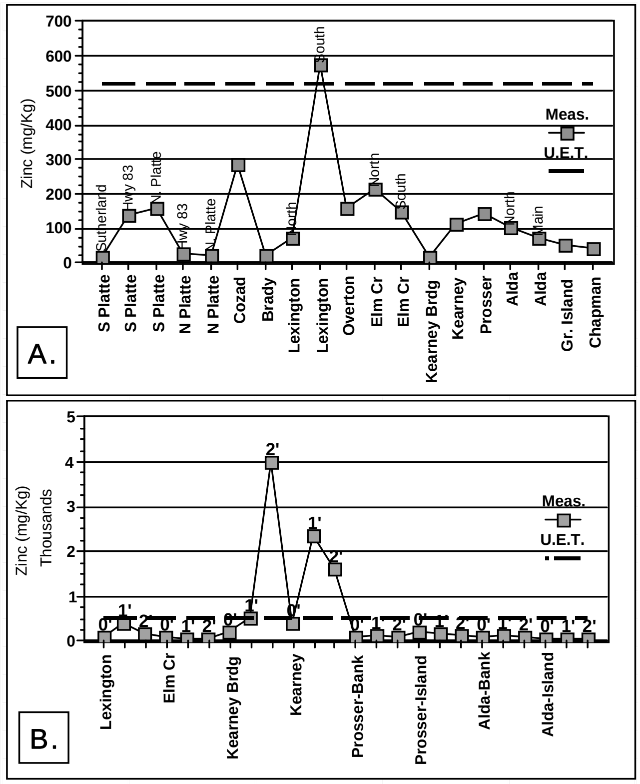


Table 169. Zinc concentrations in bed (A.) and bank and island (B.) sediment samples from the Platte River basin in Nebraska

Table 68. Pearson correlations among copper, nickel, lead, and zinc in bed and bank sediments

Source	Variable		Copper	Nickel	Lead
Bed	Nickel	r	0.7602		
		Prob.>r	0.000158		
		n	19		
	Lead	r	0.6625	0.7522	
		Prob.>r	0.001995	0.000203	
		n	19	19	
	Zinc	r	0.9793	0.7542	0.7069
		Prob.>r	< 0.000001	0.000191	0.000714
		n	19	19	19
Bank—Island	Nickel	r	0.8870		
		Prob.>r	< 0.000001		
		n	24		
	Lead	r	0.9259	0.8930	
		Prob.>r	< 0.000001	< 0.000001	
		n	23	23	
	Zinc	r	0.9683	0.8343	0.9057
		Prob.>r	< 0.000001	< 0.000001	< 0.000001
		n	24	24	23

Governance Committee Alternative

Copper (Cu) is currently well above the UET in the bed sediments (Present), and it is projected to increase at the farther downstream sites in the habitat reach of the Platte River, but decrease in the upstream sites with implementation of the Governance Committee Alternative (Table 69). There is little change in the copper concentration at the end of the first increment at any of the sites, except of the farthest downstream site near Alda, where the copper concentration is projected to increase by about a factor of 3. At the cross-section at Lexington, where there is a projected decrease, and at Kearney Bridge, where there is a projected increase, the changes remain small at the end of the entire 61-year period used in the SEDVEG model. At the Elm Creek Bridge site, there is a projected decrease to nearly ½ the initial concentration. At the site near Alda, there is a projected further increase to more than 6-times the initial concentration at the end of the 61-year period of the SEDVEG model.

Table 69. Governance Committee Alternative — sediment metals concentrations (all in ppm)

Cross-Section	Period	Cu	Ni	Pb	Zn
Lexington - XS 3	Present	1265	6.8	18.8	235
	1st Increment	1264	6.8	18.7	235
	Total 1st	1113	8.9	18.2	214
Elm Cr. Bridge - XS 19	Present	1174	4.1	11.3	206
	1st Increment	1162	4.1	11.3	204
	Total 1st	685	4.9	9.3	130
Kearney Bridge - XS 22	Present	780	2.1	9.4	113
	1st Increment	789	3.3	10.9	126
	Total 1st	803	5.0	13.1	146
near Alda - XS 56	Present	412	3.1	13.0	73
	1st Increment	1358	4.6	15.3	79
	Total 1st	2562	6.5	18.4	87

If the UET concept is correct, there should be biological effects at the current concentrations of copper in the sediments. Whether there are adverse biological effects currently is not known for certain. The fish tissue data evaluated in the Special Studies section of this Appendix do not show elevated copper in fish tissue, although the number of samples was relatively small (n = 7) and may not be representative of overall conditions. Similarly, none of the bird egg samples showed elevated tissue concentrations of copper in the Platte Basin. These results would indicate that copper is not a concern from a toxicological perspective at current concentrations in the sediments. However, copper is known to cause numerous sublethal effects on enzymatic activity, energetics, and behavior in fish [see Heath (1995) for specifics]. These effects may only be expressed at the level of the individual, when a fish is under stress. Alternatively the effect may be expressed as an increased susceptibility to predation, which may have implications at the level of the population. In either case, lethality would tend to go unnoticed, but may be manifested as a reduction in populations or changes in the species composition of the fish community.

The probability of toxicity associated with copper in the sediments agrees with the UET comparison results in that it is very high (Table 70). Although there are some relatively large changes in the copper concentrations at two of the cross-sections, *i.e.* Elm Creek Bridge and near Alda, the resulting changes in the probability of toxicity is relatively small. As was noted in the Methods section above, the probability-concentration relationships are based on logistic

Table 70. Governance Committee Alternative — probability of sediment metals toxicity					
Cross-Section	Period	Cu	Ni	Pb	Zn
Lexington - XS 3	Present	0.965	0.041	0.251	0.485
	1st Increment	0.965	0.041	0.251	0.485
	Total 1st	0.959	0.056	0.244	0.451
Elm Cr. Bridge - XS 19	Present	0.961	0.023	0.155	0.438
	1st Increment	0.961	0.023	0.154	0.435
	Total 1st	0.926	0.028	0.126	0.285
Kearney Bridge - XS 22	Present	0.937	0.010	0.128	0.245
	1st Increment	0.937	0.017	0.148	0.276
	Total 1st	0.939	0.028	0.179	0.321
near Alda - XS 56	Present	0.868	0.016	0.177	0.148
	1st Increment	0.968	0.026	0.209	0.163
	Total 1st	0.985	0.039	0.247	0.181

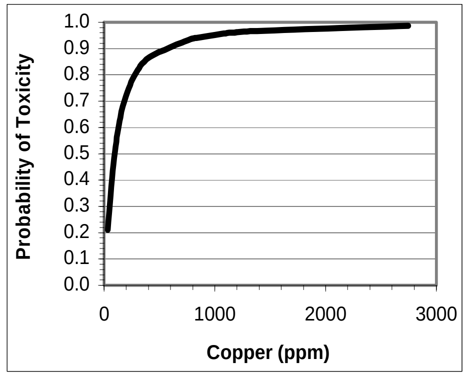


Table 170. EPA logistic curve of the probability of toxicity associated with increasing copper concentrations in sediments

regression relationships. The logistic regression for copper toxicity is shown on Table 170. The curve indicates that once the copper concentrations exceeds about 600 ppm, the probability of toxicity exceeds 0.9. Further increases in copper above that concentration cannot lead to large increases in the probability of toxicity associated with copper in the sediments. This relationship is the reason for selecting the logistic model to define the probabilities associated with high concentrations of contaminants.

The largest projected increase in the copper concentration in the sediments under the Governance Committee Alternative is at the cross-section near Alda. The concentration is projected to increase from about 400 ppm to over 2,500 ppm. The probabilities of toxicity associated with these concentrations show an increase from 0.87 under the Present Condition to 0.98 at the end of the SEDVEG model period under the Governance Committee Alternative (Table 70). Although the initial probability of toxicity is relatively high, the final probability approaches a near certainty. As was noted above in the discussion of the UET, adverse effects on aquatic life should be occurring under the Present Condition. How much the increase in copper would change these effects is difficult to predict, in that the existing effects, if any, are difficult to define.

The nickel (Ni) concentrations in the sediments under the Present Condition are greatest at the upstream cross-section at Lexington (Table 70). Nickel concentrations in the sediments decrease somewhat at each of the next two cross-sections downstream, but increase at the farthest downstream cross-section near Alda. Under the Governance Committee Alternative, nickel concentrations are not projected to change after the first increment at the Lexington and Elm Creek Bridge cross-sections, but are projected to increase at the two downstream cross-sections. At the end of the 61-year model period of the SEDVEG model, the nickel concentrations in the sediment are projected to increase at all four cross-sections, but would remain the highest at the Lexington cross-section (Table 70).

The probability of toxicity related to the nickel concentrations in the sediments is relatively low at all sites under the Present Condition (Table 70). The maximum probability of toxicity related to nickel in the sediments under the Present Condition is 0.04 at the Lexington cross-section. At that cross-section, the probability is projected to remain unchanged following the first increment, and then increase by the end of the full 61-year simulation period of the SEDVEG model (Table 70). The projected probability at the end of the full study period at the Lexington cross-section is projected to be 0.06 (or 6 chances in 100).

Nickel concentrations in the sediments are relatively low at each of the cross-sections. Nickel was carried through the analysis, because the maximum nickel concentration in the sediments exceeded the UET in Table 65. However, as is shown on Table 167, the maximum is the only sample that exceeded the UET. In addition, the sample that exceeded the UET was from a 2-foot depth bank sample. This would be included as part of the sediment that could be used for augmentation or could enter the river through bank erosion, with or without the Program. The sample is not included as part of the analysis, because the location is not among those currently being considered as source material for sediment augmentation.

Lead (Pb), like the previous two metals, shows its highest concentration at the Lexington cross-section under the Present Condition (Table 70). The lead concentration decreases at the next two cross-section downstream, at which point the concentration is ½ what it had been at Lexington. The lead concentration increases again at the cross-section near Alda. Sediment augmentation under the Governance Committee Alternative is projected to cause little change in the lead concentration at the Lexington cross-section, a small decrease (2 ppm) at the Elm Creek Bridge cross-section, and increases at the farthest downstream cross-sections. The increase in the lead concentration at the cross-section would be sufficient that at the end of the 61-year SEDVEG simulation period, the highest lead concentration would be at the cross-section near Alda, although only slightly so (Table 70).

The greatest probability of toxicity associated with lead in the sediments under the Present Condition is around 0.25 (or 1 in 4 – Table 70). Under the Present Condition, the highest probability is at the Lexington cross-section. The probability of toxicity due to lead is not projected to change greatly at the site near Lexington. The greatest change would be the increase at the Alda cross-section, where the probability of toxicity would increase to about 0.25 at the end of the 61-year SEDVEG simulation period. At the other two cross-sections, the probability of toxicity is below 0.2 in the Present Condition and remains there until the end of the simulation period. Sediment augmentation under the Governance Committee Alternative would not change the probability of toxicity due to lead to any great extent (Table 70).

The zinc (Zn) concentration in the Platte river sediments shows a distinct decreasing trend in a downstream direction (Table 70). In the Present Condition, zinc decreases steadily for 235 ppm at the Lexington cross-section to 73 ppm in the cross-section near Alda. The concentration at the Kearney Bridge cross-section is about ½ what it had been near Lexington. The probability of toxicity decreases at the more upstream cross-sections and increases at the two downstream cross-sections with implementation of the Governance Committee Alternative. At the end of the 61-year SEDVEG simulation period, the decreasing trend in potential toxicity with distance downstream is projected to remain, from about 0.5 at the Lexington cross-section to 0.3 at the Elm Creek Bridge and Kearney Bridge sections to 0.2 near Alda. None of the final probabilities of toxicity associated with zinc in the sediments would represent a significant increase relative to those of the Present Condition.

Water Emphasis Alternative

Table 71 shows the effects on concentrations of copper, nickel, lead, and zinc with implementation of the Water Emphasis Alternative. The changes in the concentrations of the four metals in the sediments under the Water Emphasis Alternative show patterns very much like those of the Governance Committee Alternative, but the end points tend to be different by several parts per million or less for three of the four metals (compare tables 73 and 75). The exception is for copper at the Lexington, Elm Creek Bridge and near Alda cross-sections where the changes may be as much as 50 ppm less under the Water Emphasis Alternative. Alternatively, the increase in sediment copper concentrations at the Kearney Bridge cross-section is virtually the same under the Water Emphasis and Governance Committee alternatives, but there is a large difference in the zinc concentrations between the two alternatives. Table 72 shows the probabilities of toxicity associated with the metals concentrations in Table 71. Once again, because of the area of the copper concentration-toxicity curve on which the sediment data reside, there is little difference in the copper toxicities between the results for the Water Emphasis and Governance Committee alternatives. The large decrease in the zinc concentration at the Elm Creek Bridge cross-section translates into the largest change in the probability

Table 71. Water Emphasis Alternative— sediment metals concentrations (all in ppm)

Cross-Section	Period	Cu	Ni	Pb	Zn
Lexington	Present	1265	6.8	18.8	235
- XS 3	1st Increment	1265	6.8	18.8	235
	Total 1st	1124	8.7	18.2	216
Elm Cr. Bridge	Present	1174	4.1	11.3	206
- XS 19	1st Increment	1174	4.1	11.3	206
	Total 1st	753	4.8	9.5	141
Kearney Bridge	Present	780	2.1	9.4	113
- XS 22	1st Increment	789	3.3	10.9	127
	Total 1st	805	5.2	13.4	149
near Alda	Present	412	3.1	13.0	73
- XS 56	1st Increment	1300	4.5	15.2	79
	Total 1st	2577	6.5	18.4	87

Table 72. Water Emphasis Alternative — probability of sediment toxicity due to four metals

Cross-Section	Period	Cu	Ni	Pb	Zn
Lexington	Present	0.965	0.041	0.251	0.485
- XS 3	1st Increment	0.965	0.041	0.251	0.485
	Total 1st	0.959	0.055	0.245	0.454
Elm Cr. Bridge	Present	0.961	0.023	0.155	0.438
- XS 19	1st Increment	0.961	0.023	0.155	0.438
	Total 1st	0.934	0.027	0.130	0.309
Kearney Bridge	Present	0.937	0.010	0.128	0.245
- XS 22	1st Increment	0.938	0.018	0.150	0.278
	Total 1st	0.939	0.030	0.183	0.327
near Alda	Present	0.868	0.016	0.177	0.148
- XS 56	1st Increment	0.966	0.025	0.207	0.162
	Total 1st	0.985	0.039	0.247	0.181

of toxicity for any of the metals at any of the cross-sections under the Water Emphasis Alternative. However, the probabilities of toxicity increase due to zinc increase at the Kearney Bridge and Alda cross-section, but the highest probability associated with zinc remains at the Lexington cross-section. Relative to the copper probability of toxicity, those of the other three metals appear inconsequential.

Table 73 shows the projected metals concentrations in the Platte River bed sediments with implementation of the Full Water Leasing Alternative. In other areas affected by the Program, the effects of the Full Water Leasing Alternative were completely different from those of the other action alternatives, with the Governance Committee Alternative used as the most detailed example. In the Middle Platte, the goal of the Full Water Leasing Alternative, including sediment augmentation, is the same as the other action alternatives. As a consequence, the changes in the sediment metals concentrations are also very much like those of the other action alternatives, as can be seen by comparisons of tables 73 and 75 with Table 73.

Table 73. Full Water Leasing Alternative — sediment metals concentrations (all in ppm)

Cross-Section	Period	Cu	Ni	Pb	Zn
Lexington	Present	1265	6.8	18.8	235
- XS 3	1st Increment	1265	6.8	18.8	235
	Total 1st	1126	8.7	18.2	216
Elm Cr. Bridge	Present	1174	4.1	11.3	206
- XS 19	1st Increment	1174	4.1	11.3	206
	Total 1st	735	4.8	9.5	138
Kearney Bridge	Present	780	2.1	9.4	113
- XS 22	1st Increment	788	3.1	10.7	125
	Total 1st	802	4.8	12.9	145
near Alda	Present	412	3.1	13.0	73
- XS 56	1st Increment	1293	4.5	15.2	79
	Total 1st	2581	6.5	18.4	87

Table 74 shows the probability of toxicity associated with the metals concentrations for the Full Water Leasing Alternative that were presented in Table 73 above. If the metals concentrations in the Platte River bed sediments are similar among the other action alternatives and the Full Water Leasing Alternative, then the associated probabilities of toxicity should also be similar. This can be verified by comparing tables 74 and 76 with Table 74.

Table 74. Full Water Leasing Alternative — probability of sediment toxicity

Cross-Section	Period	Cu	Ni	Pb	Zn
Lexington	Present	0.965	0.041	0.251	0.485
- XS 3	1st Increment	0.965	0.041	0.251	0.485
	Total 1st	0.959	0.054	0.245	0.454
Elm Cr. Bridge	Present	0.961	0.023	0.155	0.438
- XS 19	1st Increment	0.961	0.023	0.155	0.438
	Total 1st	0.932	0.028	0.129	0.303
Kearney Bridge	Present	0.937	0.010	0.128	0.245
- XS 22	1st Increment	0.937	0.017	0.146	0.273
	Total 1st	0.939	0.028	0.177	0.318
near Alda	Present	0.868	0.016	0.177	0.148
- XS 56	1st Increment	0.966	0.025	0.207	0.162
	Total 1st	0.985	0.039	0.248	0.181

Table 75 shows both the metals concentrations and their associated probabilities of toxicity for the Wet Meadow Emphasis Alternative at each of the four cross-sections in the habitat reach of the Middle Platte River. The results shown in Table 75 for the Wet Meadow Emphasis Alternative can be compared with the previous results the other three action alternatives. It should be obvious by now, that the various alternatives are not going to show great differences in either metals concentrations or probabilities of toxicity among the alternatives. The main reason for this is that all of the alternatives have the goals of providing a flow regime in the Middle Platte River that will favor the endangered species, along with a flow and sediment augmentation regime that will preserve and improve the riverine habitat conditions for those species. As a result, the flows and sediment transport are going to be similar enough among the alternatives that the metals concentrations in the sediments are going to be similar among those alternatives.

Table 75. Wet Meadow Emphasis — sediment metals concentrations and probability of toxicity

Cross-Section	Period	Metals Concentration (ppm)				Probability of Toxicity			
		Cu	Ni	Pb	Zn	Cu	Ni	Pb	Zn
Lexington	Initial	1265	6.8	18.8	235	0.965	0.041	0.251	0.485
- XS 3	1st Increment	1265	6.8	18.8	235	0.965	0.041	0.251	0.485
	Total 1st	1120	8.8	18.2	215	0.959	0.055	0.245	0.453
Elm Cr. Bridge	Initial	1174	4.1	11.3	206	0.961	0.023	0.155	0.438
- XS 19	1st Increment	1174	4.1	11.3	206	0.961	0.023	0.155	0.438
	Total 1st	748	4.8	9.5	140	0.933	0.027	0.130	0.307
Kearney Bridge	Initial	780	2.1	9.4	113	0.937	0.010	0.128	0.245
- XS 22	1st Increment	790	3.3	11.0	127	0.938	0.018	0.150	0.278
	Total 1st	804	5.1	13.3	148	0.939	0.029	0.182	0.325
near Alda	Initial	412	3.1	13.0	73	0.868	0.016	0.177	0.148
- XS 56	1st Increment	1536	4.9	15.8	80	0.972	0.028	0.215	0.165
	Total 1st	2633	6.6	18.6	87	0.986	0.040	0.249	0.182

Platte River Basin Water Quality – Impaired Waters

The Platte River is formed at the confluence of the North Platte and South Platte rivers at North Platte, Nebraska. Both the North and South Platte rivers arise in Colorado. The North Platte River flows north through Wyoming into Nebraska, while the South Platte River flows east through Colorado to the confluence in Nebraska. From a water quality jurisdictional perspective, each of the 3 States is responsible for parts of the basin that could be affected by the Program. This section of this report will focus on the lists of impaired waters [Clean Water Act, Section 303(d) lists] prepared by each of the States and approved by the Environmental Protection Agency (EPA). Under the requirements of the Clean Water Act, total maximum daily loads (TMDLs) must be developed for impaired waters in order to assure that the classified stream reaches meet their assigned water quality standards, or in terms of the Clean Water Act, support their assigned beneficial uses. Because of the emphasis that the Clean Water Act places on impaired waters, the basis for listing these streams will be given additional consideration in this FEIS appendix.

South Platte River

Colorado

The program in Colorado involves the Tamarack element. The Tamarack diversion point is located near Crook, about 35 miles from the Colorado-Nebraska State Line. However, water leasing in Colorado could involve various reservoirs in the South Platte Basin. The potential source reservoirs are all off-stream. The farthest upstream reservoir on the list of potential sources of leased water is Riverside Reservoir. In addition, direct diversions from the river throughout the basin are also included as potential sources of leased water. For this reason, the South Platte River in Colorado from near Wiggins (west of Fort Morgan) to the State Line is potentially affected.

The current 303(d) list for Colorado was prepared in April 2004 (WQCD, 2004). Several reaches of the South Platte River are listed as impaired, but all of these are well upstream from the reach of the river potentially affected by the Program. Subsequent to the preparation of the 303(d) list by the State, EPA added segments during their review of the State list. The reach of the river from Big Dry Creek to State Highway 60 was added by EPA for low dissolved oxygen. That reach is also upstream from the reach of the South Platte River potentially affected by the Program.

Colorado has a secondary list of waters that are suspected of impairment, but the existing data are either insufficient or of inadequate quality to support the listing (WQCD, 2004). This list is labeled the monitoring and evaluation (M & E) list. The reach of the South Platte River from the Weld-Morgan county line to the State Line is on the M & E list for suspected non-support of the aquatic life classified use, although the basis for the nonsupport is not specified. Prewitt Reservoir, which is included on the list of potential water sources for the Program, is also on the M & E list. Because the cause of the potential impairment is unknown, no further evaluation of the effects of the Program on the South Platte River, beyond what has been presented for the Tamarack Element, will be presented here.

Nebraska

There are several reaches of the South Platte River in Nebraska on the current State 303(d) list (NDEQ, 2004). The constituents of concern include PCBs (polychlorinated biphenyls), *E. coli*, pH, and selenium. Two of the 3 reaches listed for PCBs are in the canal between Sutherland Reservoir and Lake Maloney and between Lake Maloney and the return to the river. The other reach is located in the river upstream from the Sutherland Diversion. Sutherland Reservoir itself is also listed as impaired because of PCBs. The reaches listed for selenium are near the State Line and near the Keith-Lincoln county line.

The reach of the South Platte River from the Western Canal to the Korty Diversion Dam (subbasin SP1-70000) is listed for noncompliance with the pH standard. The pH standard may not be met if the pH is either too high or too low, *i.e.* to meet the standard, the pH must be between 6.5 and 9.0. The most common causes of low pH are industrial discharges and acid mine drainage, neither of which is occurring in the listed reach. Alternatively, high pH can be caused by instream photosynthesis due to excessive algal or plant growth, which is apparently the case in the South Platte River (Patrick O'Brien, Nebraska Department of Environmental Quality, Lincoln, Nebraska, personal communication to Jim Yahnke, Bureau of Reclamation, Denver, Colorado, November 3, 2005). Such a cause of high pH is often accompanied by low dissolved oxygen (DO) at night when photosynthesis stops and the algae or plants respire. The actual cause of the high pH is unknown (*ibid.*).

Data on contaminants in the South Platte River in Nebraska were retrieved from the EPA STORET database. Fish tissue data collected by EPA Region 7 were included in the retrieval. Among the analytes in the EPA data were several PCBs. These are shown on Figure 171. Arochlor was the commercial name of a number of PCB mixtures that were primarily used in electrical equipment, such as transformers and capacitors. The more common formulations include:

PCB-1221 (Arochlor 1221)
PCB-1232 (Arochlor 1232)
PCB-1242 (Arochlor 1242)
PCB-1248 (Arochlor 1248)
PCB-1254 (Arochlor 1254)
PCB-1260 (Arochlor 1260)
PCB-1016 (Arochlor 1016).

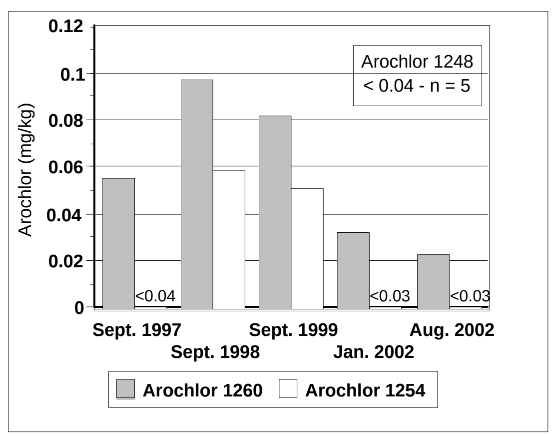


Figure 171. South Platte River at Paxton – fish tissue PCB concentrations

(Source: http://www.epa.gov/owow/oceans/regulatory/sec301tech/table2.html, accessed on Nov. 7, 2005.) The first two numbers associated with the PCBs refer the type of mixture and the second two numbers refer to the percent chlorine in the PCB compounds, *e.g.* Arochlor 1254 is 54 percent chlorine by weight (ATSDR, 2000). The manufacture of PCBs was banned in 1977, but because of their persistence, high food chain concentrations persist into the present.

Of the 3 Arochlor mixtures shown on Figure 171, Arochlor 1248 was not detected at the reporting limit of 0.04 mg/kg (= ppm), while Arochlor 1254 was detected in 2 of the 5 samples in which it was analyzed. As is indicated on Figure 171, the reporting limit varied among the Arochlor 1254 samples. Alternatively, Arochlor 1260 was detectable in all 5 samples in which it was analyzed. The lower 3 concentrations of Arochlor 1260 coincided with the sample dates on which Arochlor 1254 was below the reporting limit, while the highest 2 concentrations of Arochlor 1260 coincided with the samples in which Arochlor 1254 was present at detectable concentrations (Figure 171). These results may indicate some contagion (*e.g.* common source) between the 2 Arochlor mixtures. Based on information presented in Niimi (1996), the observed PCB concentrations in fish tissue shown on Figure 172 would be below toxic levels to fish.

Total selenium monitoring data from 2002 and 2003 on the South Platte River at 2 sites in Nebraska were provided by the NDEQ (Dave Ihrie, Nebraska Department of Environmental Control, Lincoln, Nebraska, personal communication, email of December 5, 2005 to Jim Yahnke, Bureau of Reclamation, Denver). The data are plotted on Figure 172, which also shows the State selenium standard. The State standard is 5 µg/L, which is also the reporting limit for the samples. Consequently, any time there is measurable selenium, it will either just meet or exceed the standard in the samples.

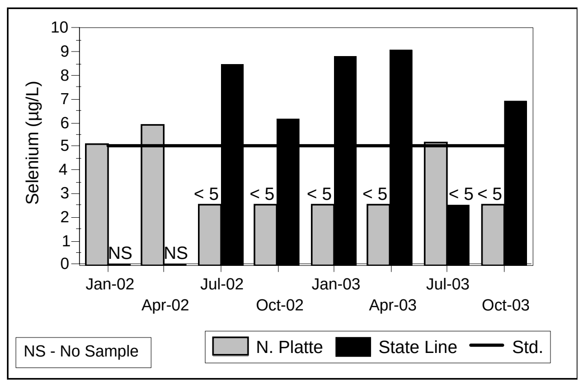


Figure 172. Total selenium at 2 sites on the South Platte River, Nebraska

The State Line site is located near the Julesburg gage. Selenium exceeded the standard in all but 1 of the 6 State Line samples shown on Figure 172. Meanwhile, the majority of the selenium samples at the downstream site near North Platte were below the detection limit and 2 of the samples in which selenium was measurable were essentially at the detection limit (Figure 172). The April 2002 sample did, however, exceed the selenium standard.

The EPA samples described above for PCBs were also analyzed for selenium. The selenium concentrations in the fish are shown on Figure 173. There are 2 concentrations flagged on Figure 173. The upper level is above 4 mg/kg, which is considered the toxicity threshold of the NIWQP (1998). The level between 3 and 4 mg/kg is labeled the level of concern, which is intermediate between the toxicity threshold and the no effect level for warm water fish (*ibid.*).

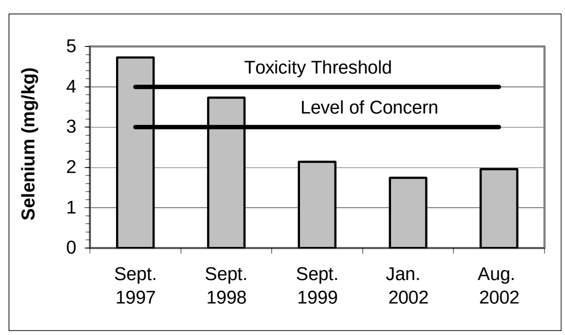


Figure 173. Selenium concentrations in fish tissue – South Platte River at Paxton, Nebraska

The level of concern is where some physiological effects to sensitive species or individuals could occur.

The sample collected in 1997 shows a selenium concentration above the toxicity threshold. The 1998 sample shows a concentration within the level of concern. The subsequent samples collected in 1999 and 2002 show selenium concentrations below the no effect level. Whether the decreases over time indicate a decreasing trend is difficult to verify from the data, but such a trend is indicated. Future data would be needed to confirm whether or not there is a trend.

North Platte River

Wyoming

The Department of Environmental Quality (Wyoming DEQ, 2004) recently completed their biennial review of water quality in Wyoming. Most of the impaired waters, *i.e.* those that do not meet water quality standards for designated uses, consist of tributaries that are not included within the potentially affected environment. However, the reach of the mainstem of the North Platte River in the vicinity of Casper is listed as impaired due to excessive concentrations of selenium in return flows from the Kendrick Project. The Kendrick Project selenium problem was studied extensively by the Department of the Interior's National Irrigation Water Quality Program (NIWQP) from 1986 though 2002 (Peterson *et al.*, 1988; See *et al.*, 1992; NIWQP, 2004).

The NIWQP collected data on selenium concentrations in the mainstem of the river and a number of tributaries draining the Kendrick Project between 1986 and 1995. In addition there are periodic USGS monitoring data available for other tributaries and the mainstem of the river. The data are summarized on Figure 174. As can be seen, the selenium concentrations in the tributaries (Figure 174B) dwarf those of the mainstem of the North Platte River (Figure 174A). The selenium concentrations in the tributaries are in the tens to hundreds of µg/L, although the medians are less than 100 µg/L (Figure 174B); because of the wide range in selenium concentrations in the tributaries, the *y*-axis on Figure 174B is plotted on a logarithmic scale. Alternatively, the median selenium concentrations in the mainstem of the North Platte River are all less than the selenium standard of 5 µg/L, as indicated by the *y*-axis on Figure 174A.

Figure 174 shows plots of both dissolved and total selenium. The plots of the mainstem sites indicate that the median dissolved and total selenium are essentially equal at all but 1 of the sites, the one at Mills, where there is a 1 µg/L difference (Figure 174A). In other words, essentially all of the selenium is dissolved in the mainstem of the North Platte River, at least on most occasions.

The dissolved and total selenium are also about the same in most of the tributaries (Figure 174B). There are no total selenium data from Lonetree Creek. The median dissolved and total selenium are equal in the Sweetwater River (site S-water), Poison Spring Creek upstream from the Kendrick Unit (P. Spring1), and both sites on Poison Spider Creek (P. Spider 1 & 2). Actually, both total and dissolved selenium are below the detection limit at the upstream site on Poison Spring Creek. Total selenium is 1 or 2 µg/L greater than the dissolved fraction in the Medicine Bow River and Bates Creek.

The remaining 3 sites show a much different pattern between total and dissolved selenium from the other 6 sites where there are both total and dissolved selenium data. In Casper Creek, the total selenium is about 20 µg/L greater than the dissolved fraction. This would indicate that there is a significant amount of particulate selenium in Casper Creek. What the composition of the particulate fraction could be is unknown, but it may consist of adsorbed selenite, particulate organic selenium, or even eroded mineral selenium, the most common of which is elemental selenium.

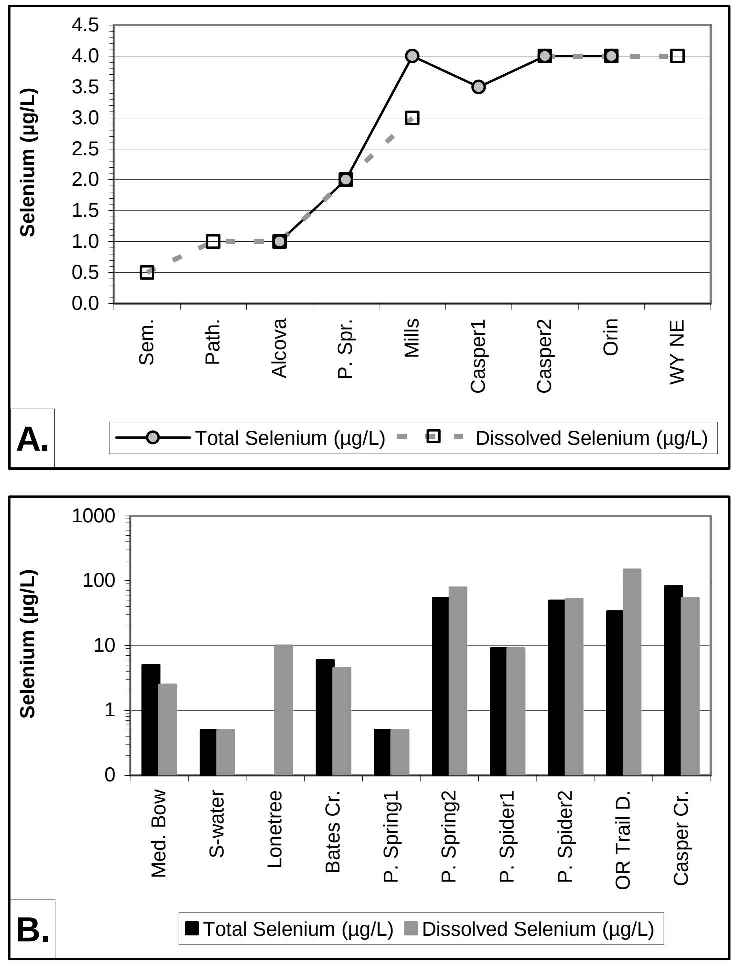


Figure 174. Median selenium concentration in the North Platte River: A. Mainstem B. Tributaries

The last 2 sites (Poison Spring near the mouth (P. Spring2) and the Oregon Trail Drain (OR Trail D.) represent something of an anomaly in that the median dissolved fraction is greater than the total. Since it is physically impossible for the dissolved fraction to be greater than the total selenium, the difference must be due to some factor in the data. Actually, the difference reflects a change in the sampling philosophy on the part of the USGS. In the early part of the record, the USGS collected samples for both dissolved and total selenium analysis. Even earlier, only total selenium was analyzed. In the later part of the record in the NIWQP studies, only the dissolved fraction was analyzed. The difference represents somewhat higher selenium concentrations in the later part of the study period and includes the much larger body of data. This does not mean that the selenium concentration was necessarily higher, *i.e.* an increasing trend, but only that the dissolved selenium samples encompassed a wider range of conditions, in particular, drought.

Figure 175 shows minimum, median and maximum selenium concentrations at sites on the mainstem of the North Platte River in comparison to the water quality standard for selenium. Although the standard is met most of the time, there are obviously times during which the standard is exceeded by at least a factor of 2 at the monitoring sites downstream from Casper and

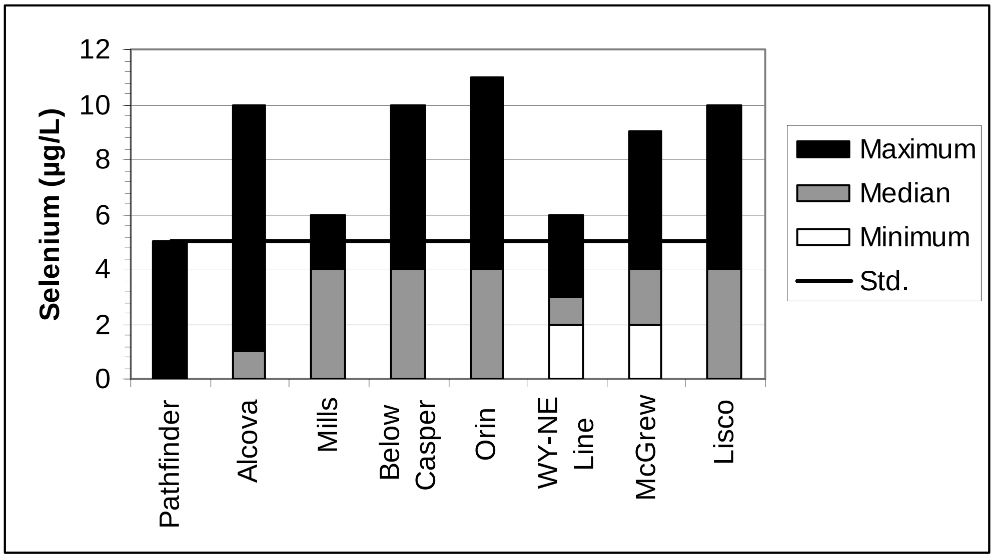


Figure 175. Comparison of selenium concentrations in the North Platte River from above Pathfinder Reservoir in Wyoming to Lisco, Nebraska, to its water quality standard

the one at Orin. The long-term maximum selenium concentration at the State Line is probably the same as those below Casper and at Orin, but there are only 3 samples from that site. These samples may or may not be representative of the long-term selenium distribution. The selenium concentrations at McGrew and Lisco in Nebraska are similar to those at the sites below Casper and at Orin in Wyoming.

The Orin gage is just upstream from Glendo Reservoir, which would be expected to have an effect on the selenium concentration. In addition, Guernsey Reservoir and the Laramie River would also have some effect on the downstream selenium concentration between Glendo Dam and the State Line gage. The reservoirs would tend to decrease selenium because of biological uptake and settling, while the Laramie River would be a source of dilution (maximum observed selenium of 1 µg/L). Because of these factors, the rather limited data for the State Line gage may actually be representative of the longer-term statistical selenium distribution. Samples collected by Druliner *et al.* (1999) at Whalen Diversion Dam had selenium concentrations the same range as those at the State Line, *i.e.* 3-4 µg/L, further indicating that the State Line data may be representative samples.

Although the maximum selenium concentrations vary considerably, the medians show a general increase in a downstream direction. Beginning with the site above Pathfinder, where the median selenium concentration is below the detection limit, there is an increase to 1 µg/L at Alcova (downstream from Alcova Dam), and a further increase to 4 µg/L at Mills. From Mills downstream, the median selenium concentration remains at about 4 µg/L (Figures 174A and 175).

Although the North Platte River in the vicinity of Casper and Glenrock, Wyoming, is listed as impaired for selenium, there are no plans for developing a TMDL for the river and its tributaries (Wyoming Department of Environmental Quality [DEQ, 2004). Because of ongoing efforts by the irrigation district, conservation district, and Natural Resources Conservation Service to control selenium on the irrigated lands, the river reach has be given a low priority for TMDL development (Wyoming DEQ, 2004), because the ongoing program is essentially what would be required if a TMDL were developed. There are currently no other listed sources of water quality impairment of the North Platte mainstem in Wyoming.

Nebraska

The North Platte River in Nebraska has several mainstem reaches on its current (2004) list of impaired waters (NDEQ, 2004). The pollutants causing the impairment or the parameters identified as impaired are: *E. coli* (2), mercury (2), dieldrin (1), temperature (1), and PCBs (NDEQ, 2004). Numbers in parentheses indicate the number of listed reaches. TMDLs have been developed for fecal coliform bacteria (NDEQ, 2003a). The TMDL is applicable to all bacterial impairments, including *E. coli.*

The mercury impairment is included on the basis on a fish consumption advisory. No TMDL is contemplated for mercury, because the State feels that the source is airborne rather than a water quality problem. The dieldrin and PCB listings are also based on fish tissue data. However, neither dieldrin not PCBs are currently being used, and the source is unknown (NDEQ, 2004). The most likely source is contamination of sediments remaining from the time when the use of both was prevalent. Both dieldrin and PCBs are persistent chlorinated hydrocarbons, and an alternative factor may simply be continued cycling through the food chain.

Although Figure 175 indicates that selenium exceeds its aquatic life standard in the North Platte River in Nebraska, the data reflect concentrations in the river over a long period of record. More recent data of Druliner *et al.* (1999) at 2 sites between the State Line and Lake McConaughy would indicate that the selenium standard is currently being met in the North Platte River in the Nebraska panhandle. In assessing compliance with water quality standards, only recent data, *i.e.* samples collected within the last 5 years, are used. Quarterly samples collected from the North Platte River by the NDEQ in 2003 at the State Line, Bridgeport, Lewellen, and North Platte had selenium concentrations below the water quality standard of 5 µg/L.

Middle Platte Basin, Nebraska

The Middle Platte River in Nebraska has 7 reaches listed as impaired (NDEQ, 2004). These include 4 reaches for which TMDLs have been developed, 1 reach where the source is defined as natural, and 2 reaches for which a TMDL has not been developed [303(d) waters] (NDEQ, 2003b). The parameter identified as impaired is again *E. coli* (*ibid.*). *E. coli* is a member of the fecal coliform group.

A summary of the data from the Middle Platte fecal coliform TMDL is shown in Table 76. The criterion shown in Table 76 is 400 colonies/100 mL; this is the standard that should not be exceeded in more than 10 percent of the samples during the recreation season (NDEQ, 2004). There is also a standard based on a monthly geometric mean of 200 colonies per 100 mL. Both standards were exceeded in the same stream segments, which include MP1-10000, MP1-20000, MP2-20000 and MP2-40000. The Middle Platte TMDL for bacteria applies to those 4 of the 6 segments of the river shown in Table 76. Only the segments in subbasin MP2, the downstream end of which is near Grand Island, are likely to be affected by the Program. Subbasin MP-1 is east of Grand Island and is not included in the Program hydrology model, but increases in flow in the subbasin are expected, although they should be small relative to normal river flows, which increase greatly downstream from the mouth of the Loup River.

Table 76. Middle Platte River – 2001 assessed fecal coliform data and assessments

Segment	**Site Location**	**USGS/DNR Gage Associated with Site**	**Number of Samples**	**Season Geometric Mean (#/100 ml)**	**Number Samples >400/100 ml**	**% Samples >400/100 ml**
MP1-10000	Platte River East of Columbus	None	17	397	8	47%
MP1-20000	Platte River at Duncan	06774000	22	211.1	7	32%
MP2-10000	Platte River at Grand Island	06770500	21	101.6	2	9.5%
MP2-20000	Platte River at Kearney	06770200	20	650.7	12	60%
MP2-30000	Platte River at Cozad	06766500	21	88.3	1	4.8%
MP2-40000	Platte River at Brady	06766000	21	115.8	4	19%

Source – NDEQ (2003b)

The river reach at Grand island is also listed as impaired for excessive temperature. At present, no TMDL is contemplated. The NDEQ feels that the excessive temperature is natural (Patrick O'Brien, Nebraska Department of Environmental Quality, Lincoln, Nebraska, personal communication to Jim Yahnke, Bureau of Reclamation, Denver, Colorado, November 3, 2005). The effect of the Program on the temperature of the Middle Platte River near Grand Island was evaluated in considerable detail earlier in this appendix.

Effects of Program Activities on Section 303(d) Listed Streams

In general, the Program increases flows in the affected streams in the Platte River basin, including the North and South Platte rivers. These increases generally provide dilution water that can dilute the concentration on the contaminants described above. The following sections will look at the effects of the increased flows at the points described above.

South Platte River

Flows in the South Platte River in Colorado and Nebraska could be affected by the Tamarack Project, and flows in the South Platte River in Nebraska could be affected by the change in operations of the Central Nebraska Public Power and Irrigation District facilities. The Tamarack Project consists of diversions for ground water recharge of the alluvial aquifer near Crook, Colorado. The water than returns later in the year. There are currently no 303(d) listed reaches of the South Platte River in Colorado. The effects of the Tamarack Project in Colorado will be evaluated on the basis of specific electrical conductance (EC) later in this appendix.

The 303(d) listed reach of the South Platte River in Nebraska that could be affected by the Tamarack Project is at the State Line. The reach is listed for selenium As will be shown in detail later, there is a good relationship between selenium and EC in the North Platte River in Wyoming. However, there is no significant relationship between selenium and EC in the South Platte near the State Line (Figure 176). As a consequence, the EC analysis will not necessarily define the effect on selenium in the South Platte River.

There is evidence to indicate that the selenium originates upstream from the diversion point for the Tamarack Project. Samples collected in 1993 by the USGS at a site near Balzac and downloaded from NWIS have a minimum selenium concentration (4 µg/L) that is greater than the long-term median selenium concentration at the Julesburg gage (3 µg/L). If the selenium in the South Platte River does originate upstream from the Tamarack diversion point, then the same mechanism that affects EC could apply to selenium. The water diverted from the South Platte River near Crook has a relatively high total dissolved solids (TDS), and thus, EC. EC is a surrogate measurement of TDS. The water diverted for recharge on the Tamarack site is slightly concentrated by evaporation as it sits in the recharge pits. However, this concentrating effect is more than offset by dilution from the ground water from the sandhills aquifer. As is described later in the more detailed analysis of the effects of the Tamarack operation on the EC of the ground water at the site, the enhanced recharge increases the discharge from the sandhills aquifer and reduces the EC of the discharge from the site. The same mechanism should also reduce the selenium concentration.

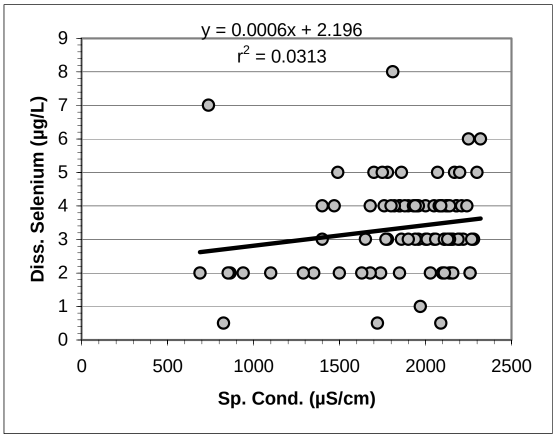


Figure 176. Relationship between dissolved selenium and EC in the South Platte River at Julesburg

Much or all of the South Platte River in Nebraska is diverted at Sutherland, where it is mixed with water released from Lake McConaughy. The operation results in dilution of the selenium, as illustrated on Figure 172. The Program will increase releases from Lake McConaughy, although not all of the releases may be diverted at Korty to mix with the South Platte diversions. There should be no reduction in the dilution; so the net effect may be somewhere between no change in the amount of water available to dilute selenium in the South Platte to an increase. In either case, there is likely to be a reduction in the selenium concentration below the Sutherland return.

North Platte River

The indicator for the effects of alternatives on the North Platte River in Wyoming is total dissolved solids (TDS). The complete analysis of those effects is presented in an earlier section of this appendix. Because selenium is the cause of impairment in the North Platte River, this section will present an overview of the potential changes in selenium between Grey Reef Dam and the Orin gage. These are the 2 sites in the North Platte EIS operations model where there are flow data for the impaired river reach. There is also a more complete description of the sources and historic effects of selenium loads in this river reach in the above North Platte section of this appendix. In addition, there is a discussion of the relationship between selenium and TDS in that section as well.

The selenium data for the tributaries are limited in comparison with the period of record for the operations model. To compensate, the tributaries were represented by their median flow and selenium concentrations. Because there are other inflows beside the monitored tributaries, the inflow and selenium concentrations were back-calculated at the median flow and selenium concentrations of the river at the Alcova (gage downstream from Gray Reef Dam) and Orin gages. The results are shown in Table 77.

Table 77. North Platte and tributary flow (ft^3/s) and selenium concentrations (µg/L)

Site	Flow	Diss. Se	Total Se
North Platte at Alcova	1000	1	1
Tributaries	40	46.1	48.9
Alcova-Orin gains	400	7.3	7.3
North Platte at Orin	1440	4	4

The river data in Table 77 were retrieved from the USGS National Water Information System (NWIS) database. The selenium concentrations are reported to the nearest µg/L. The tributary flows and concentrations are also the median flow and selenium concentrations from NWIS. In most, but not all cases, the selenium concentrations for the tributaries are also reported to the nearest µg/L. The calculated selenium concentrations represent the flow weighted average of the concentrations from the monitored tributaries. The tributaries include Lonetree, Bates, Poison Spring, Poison Spider, and Casper creeks and the Oregon Trail Drain. The selenium concentration of the ungaged gains was calculated from the data on the North Platte at Alcova and Orin, after subtracting out the contribution from the tributaries. If this analysis is at all accurate, it would indicate that there are selenium contributions from other sources in addition to the measured tributaries.

As can be seen from the data in Table 77, the inflowing selenium from the tributaries will be diluted by the river. The effect of the tributaries will be greatest during low-flows in the river. To evaluate something of a worst case condition, a low flow month at the time that irrigation return flows (the source of the selenium) would still be contributing was selected. The lowest flow month that met the above criteria is September. All of the flows used in the selenium analysis represent September flows from the North Platte EIS operations model.

The effects of the tributaries on the selenium concentrations were calculated from the data in Table 77, except that difference between the Grey Reef and Orin flows from the operations model was substituted for the median flow to estimate the load from the ungaged gains. This difference was multiplied by the selenium concentrations shown in Table 77 for the ungaged gains. However, at times, the operations model output indicated that the river reach was losing water, rather than gaining water. In these cases, the ungaged gains were assumed to be 0 and were not included in the selenium calculations for the river.

The comparison of the selenium concentrations for each of the alternatives and the Present Condition is shown in Table 78. The selenium concentrations in Table 78 represent the median and maximum concentrations calculated for each year in the operations model period of record, 1947-1994, based on September flows. In all cases, there is a decrease in the median and maximum dissolved and total selenium concentrations with all of the alternatives in comparison with the Present Condition. This is what would be expected with an increase in flow in the river. As was noted above, the main effect of the alternatives is to increase deliveries of water to the Critical Habitat area in Nebraska. In doing so, additional dilution water is provided to the selenium loadings from both the measured and unmeasured inflows to the North Platte River in Wyoming.

The North Platte River in Nebraska is listed as impaired because of excess bacterial concentrations. The indicator for the effects of the Program alternatives on the North Platte River in Nebraska is also TDS. Flows in the North Platte River in Nebraska will also increase under all of the alternatives. The additional flows of the North Platte River in Wyoming would be delivered to Lake McConaughy. The effects on TDS in the North Platte River upstream from Lake McConaughy at the Lewellen gage are analyzed in the earlier North Platte Basin section of this appendix.

Table 78. Summary of the effects of alternatives on selenium concentrations in the North Platte River between Grey Reef Dam and the Orin gage during the low-flow month of September

Alternative	Grey Reef Flow (ft^3/s)		Orin Flow (ft^3/s)		Orin Dissolved Se (µg/L)		Orin Total Se (µg/L)	
	Median	Minimum	Median	Minimum	Median	Maximum	Median	Maximum
Present Condition	523	502	712	425	4.6	5.6	4.8	5.7
Governance Committee	797	502	828	440	3.8	5.3	3.9	5.5
Water Emphasis	1458	539	1621	607	2.7	5.3	2.7	5.5
Full Water Leasing	986	551	866	555	3.6	5.5	3.8	5.6
Wet Meadow Emphasis	1141	442	1316	578	2.9	5.4	3.0	5.5

There is no direct relationship between the effects of increased flow on TDS and those on bacterial concentrations in the river. If the source of the bacteria is a point source that is constantly discharging, *e.g.* a wastewater treatment plant, than there could be additional dilution, as occurred above with selenium. Alternatively, if the main source is intermittent, *e.g.* storm water runoff, then there is unlikely to be any great effect on bacterial concentrations due to the increased flows from the Program alternatives.

NDEQ (2003a) breaks down the sources into percent bacterial load contributions to the impaired reaches of the North Platte River by nonpoint and point source totals. The percent contribution by nonpoint sources range from 65 to as much as 98 percent.

The increases in flow are probably too small to have any great dilution effect on storm water loadings, which tend to have the greatest effect on nonpoint source loadings. On the other hand, there should not be an adverse effect that would increase bacterial concentrations either.

Middle Platte River

As was done in the North Platte Basin TMDL, NDEQ (2003b) breaks down the sources into percent bacterial load contributions to the impaired reaches of the Middle Platte River by nonpoint and point source totals. The percent contribution by nonpoint sources in the Middle Platte Basin range from 76 to as much as 98 percent.

The Middle Platte Basin encompasses the critical habitat that is the focus of the Program. The Program is designed to increase flows for whooping crane habitat maintenance and stabilize flows during the nesting season of the least tern and piping plover. Either of the factors could provide dilution for bacterial loadings to the river from point or nonpoint sources, but would probably be most effective diluting point source loadings. As was the case with the North Platte Basin, there should be no adverse effect on bacterial concentrations in the river.

As was described earlier, the Program will increase flows in the Middle Platte River during the critical months of June through August when excessive temperatures occur in the Middle Platte River. The increased flows tend to inhibit warming of the river. As a consequence, the number of times that the temperature standard is exceeded will decrease in each of the 3 months.

Contaminants in Platte River Fish

The State of Nebraska maintains a network of monitoring sites where fish tissue samples are collected and analyzed for priority pollutants (NDEQ, 1996). The fish tissue data collected between 1983 and 1997 were provided to Reclamation by Mike Callum of the NDEQ (personal communication to Jim Yahnke, Bureau of Reclamation, February 3, 2000). According to NDEQ (1996), when human health is the prime consideration, fish fillets are analyzed, and when aquatic life impacts are the prime consideration, whole fish samples are analyzed. The data provided include both types of samples.

The primary concern of NDEQ program is the potential for harm to endangered birds, particularly to the least tern and piping plover during reproduction. The least tern is piscivorus. In the study area, it consumes only small fish, *i.e.* < 3.8 cm or 1.5 in. (Wilson *et al.*, 1993). None of the NDEQ samples is anywhere near that small; the smallest fish in the NDEQ data set is just under 5 inches. In order to evaluate the fish data relative to potential toxicity to breeding terns, a certain amount of extrapolation will be necessary.

Inorganic contaminants

The NDEQ samples have been analyzed for a total of 10 inorganic contaminants. The data, including the length, weight, and the number of fish per sample, are summarized in Table 79. The analyses were performed on composite samples consisting of between 1 and 11 fish, with a median sample size of 5. Not all of the contaminants are analyzed in every sample, and in certain cases there are very few analyses. The most analyses are for mercury and the fewest are for nickel. Mercury is listed as the primary inorganic contaminant of concern in Nebraska (NDEQ, 1996).

Table 79. Summary data for inorganic contaminants in Platte River basin fish (all in ppm wet weight [ww], unless otherwise noted)

	Length (in.)	Weight (lb.)	Number of fish	Arsenic	Cadmium	Chromium	Barium
Minimum	4.9	0.3	1	0.011	0.05	0.03	0.06
Median	15.7	2.0	5	0.075	0.08	0.34	1.64
Maximum	26.3	8.1	11	0.190	0.33	0.64	4.50
# of Obs.	62	62	66	14	18	18	14
	Lead	Copper	Mercury	Nickel	Selenium	Zinc	
Minimum	0.15	0.73	0.020	0.27	0.52	17.0	
Median	0.32	0.94	0.095	0.48	1.41	64.4	
Maximum	0.71	1.33	0.358	0.94	4.73	84.3	
# of Obs.	5	7	68	4	23	7	

The National Irrigation Water Quality Program (NIWQP) was formed by the Department of the Interior after wildlife experienced losses due to dead and deformed embryos and direct toxicity traced to contaminants in irrigation drain water in the Central Valley of California in 1985. The NIWQP (1998) has developed guidelines for interpreting contaminants data in a variety of life forms. The guidelines consist of a "no effect" level and a toxicity threshold; intermediate concentrations between these two are labeled as being at a "level of concern." These guidelines will be used where possible to evaluate whether there is a potential problem involving the various contaminants for which there are data in the NDEQ data set.

The NIWQP guidelines include only 7 trace elements, two of which include no data in the NDEQ data set. Alternative evaluation criteria that were used in the NIWQP were the 85th percentile of the National Contaminants Biomonitoring Program (NCBP). The 85th percentile value or lower was assumed to be normal background, while values higher than the 85th percentile were assumed to represent a potentially contaminated situation. The 85th percentile concentrations from the NCBP have been published by Schmitt and Brumbaugh (1990). These will be used to supplement the NIWQP (1998) criteria where necessary.

The "no effect levels" and toxicity thresholds used to evaluate the fish contaminants data are shown in Table 80. A summary of the comparison between the data and the evaluation criteria is also shown in Table 80.

None of the arsenic nor copper samples exceeded their respective "no effect level." Alternatively a majority of the cadmium, lead, selenium, and zinc samples exceeded the lower level of concern (Table 80A). The NCBP had 2 sites on the Platte River in the study area. The results for cadmium and selenium were above the 85th percentile in samples collected during 1978-79 and in 1980-81 (Lowe *et al.*, 1985), while several of the samples were above the 85th percentile during the 1984 sampling (Schmitt and Brumbaugh, 1990). A large percentage of the mercury analyses also exceeded the "no effect level," although none of the NCBP samples collected were above the 85th percentile concentration. The NCBP 85th percentile mercury concentration is 0.18 ppm, which is somewhat greater than the no effect level shown in Table 2A. The frequency with which the NCBP 85th percentile would be exceeded would be less than that for the "no effect level."

Table 80B shows toxicity thresholds for the elements. The toxicity thresholds shown in Table 80B for copper, mercury, and selenium are for fish, not for waterfowl, which will be discussed later. The thresholds for cadmium and zinc are for waterfowl; no thresholds for fish could be located. There is no toxicity threshold for lead, which is not included in the NIWQP (1998) guidelines. In a review of studies of lead intake by aquatic birds, Irwin *et al.* (1997) found that the majority studies focused on lead shot, followed by wastes, such as lead-base paints either in cans or on discarded materials.

Table 80. Comparison of inorganic contaminants in Platte River basin fish to levels of concern							
A. NIWQP No Effect Levels (or NCBP 85th Percentile)							
	Arsenic	Cadmium	Lead	Copper	Mercury	Selenium	Zinc
NOAEL (ppm)	0.25[1]	0.05[2]	0.22[2]	2.45[1]	0.11[1]	0.75[1]	34.2[2]
Number > NOAEL	0	16	3	0	30	18	6
Total Observations	14	18	5	7	68	23	7
% > NOAEL	0	88.9	60.0	0	44.1	78.3	85.7
B. NIWQP Toxicity thresholds (various - see notes)							
	Arsenic	Cadmium	Lead	Copper	Mercury	Selenium	Zinc
Threshold (ppm)	3[1]	0.5[3]	NA	3.325[1]	1.0[1]	1.0[1]	80[4]
Number > threshold	0	0	NA	0	0	14	0
Total Observations	14	14	5	7	68	23	7
% > threshold	0	0	NA	0	0	60.9	0.0

[1] NIWQP (1998) no observed adverse effect level (NOAEL) or toxicity threshold. NOTE - all of the levels attributed to NIWQP (1998), which is a literature review, are derived form other sources; see NIWQP (1998) for individual references.

[2] Schmitt and Brumbaugh (1990) 85th percentile concentration

[3] Toxicity threshold for waterfowl food consumption (Irwin *et al.*, 1997)

[4] Toxicity threshold for waterfowl food consumption (NIWQP, 1998 - Table 37)

Only selenium exceeds any toxicity threshold among the elements shown in Table 80B. Nevertheless the elements that exceed the NOAEL, but not the toxicity threshold, fall within the level of concern as defined by NIWQP (1998). Consequently future monitoring should continue to evaluate those elements as well as selenium.

Although only selenium is indicated as having a threat for widespread effects on fish, the real goal of this evaluation is to look at the potential toxicity to predators of fish, specifically to the least tern. For this purpose any toxicity threshold that is lower than the ones evaluated in Table 80B needs to be reviewed. Since the toxicity thresholds for cadmium and zinc shown in Table 80B are not exceeded, those will not be evaluated further. They apparently pose only a minimal threat to predatory birds.

The selenium threshold for water bird food items in NIWQP (1998) is 3-8 ppm ww. The lowest of these is 3 times the fish threshold shown in Table 80B. Consequently selenium would be expected to be somewhat less of a threat to consumers of the fish than it would be to the fish themselves. A comparison of the selenium concentrations of selenium in the fish samples to the bird food dietary threshold of 3 ppm shows that about 9 percent of the fish samples exceeded it. The selenium threshold shown above is based on reproductive impairment, which is obviously a major concern for least terns. Based on this result, selenium would have the potential to inhibit the recovery of the least tern population in the Platte River.

The bird dietary threshold for mercury is 0.10 ppm ww. This is lower than the 0.11 ppm NOAEL for fish. Consequently mercury needs to be further evaluated for potential effects on breeding birds. The comparison of the mercury concentration in the fish samples to the bird dietary mercury threshold shows that about 47 percent exceeded it. This is about the same percentage as exceeded the NOAEL for fish (Table 80A), which is no surprise since the levels of concern are about the same as well.

Neither arsenic nor copper exceeded any of the levels of concern shown in Table 80. The toxicity thresholds for bird food items for arsenic and copper are 7.5 and 200 ppm ww respectively (NIWQP, 1998). Since these are much higher than the levels of concern for the fish shown in Table 80, these elements would be considered to pose no threat to consumers of the fish.

One further factor in evaluating the potential threat to least terns due to potential toxicity of dietary levels of inorganic contaminants concerns the food items themselves. As was noted above, least terns prey on fish that are much smaller than those in the samples being evaluated. If there is a relationship between the size of the fish and the concentration of the contaminant in question, the threat may be more or less than the evaluation above would indicate. The relationship among the various contaminants and physical characteristics of the fish sampled was evaluated by correlation analysis. The results are shown in Table 81. Significant correlations are highlighted for emphasis.

Table 81. Correlations among element concentrations, physical variables, and time - trends in

Variable		Species	Length	Weight	No. of fish	Year
Species	r	——	-0.187373	-0.304958	0.002369	0.024033
	Prob. > r	——	0.144757	0.015949	0.985419	0.845758
	n	68	62	62	62	68
Sample type	r	0.280678	0.274788	0.161902	-0.238471	0.669748
	Prob. > r	0.020426	0.030661	0.208685	0.061966	< 0.000001
	n	68	62	62	62	68
Length	r	-0.187373	——	0.853971	-0.191782	0.466427
	Prob. > r	0.144757	——	< 0.000001	0.135370	0.000133
	n	62	62	62	62	62
Weight	r	-0.304958	0.853971	——	-0.246429	0.336790
	Prob. > r	0.015949	< 0.000001	——	0.053510	0.007437
	n	62	62	62	62	62
No. of fish	r	0.002369	-0.191782	-0.246429	——	-0.365753
	Prob. > r	0.985419	0.135370	0.053510	——	0.003463
	n	62	62	62	62	62
Arsenic	r	-0.422495	0.512793	0.459654	0.140189	0.303094
	Prob. > r	0.132333	0.060782	0.098209	0.632645	0.292167
	n	14	14	14	14	14
Cadmium	r	-0.192455	0.142719	0.145846	0.225909	0.618580
	Prob. > r	0.444216	0.584768	0.576479	0.383291	0.006207
	n	18	17	17	17	18
Chromium	r	0.538811	-0.130275	-0.246378	0.061859	-0.541419
	Prob. > r	0.021044	0.618232	0.340453	0.813551	0.020311
	n	18	17	17	17	18
Barium	r	No Data	-0.378885	-0.221539	-0.261998	-0.605649
	Prob. > r		0.181566	0.446554	0.365539	0.021709
	n	14	14	14	14	14
Lead	r	0.350247	0.124524	0.009284	-0.232760	0.799020
	Prob. > r	0.563345	0.841861	0.988179	0.706339	0.104837
	n	5	5	5	5	5
Copper	r	-0.578335	-0.256640	-0.153942	0.747626	0.401531
	Prob. > r	0.173779	0.623492	0.770911	0.087502	0.371935
	n	7	6	6	6	7
Mercury	r	0.298597	0.355210	0.303746	-0.090179	0.572206
	Prob. > r	0.013385	0.004611	0.016394	0.485777	< 0.000001
	n	68	62	62	62	68
Nickel	r	0.869832	-0.662147	-0.604824	0.285231	-0.438236
	Prob. > r	0.130168	0.539291	0.586487	0.815859	0.561764
	n	4	3	3	3	4

Table 81. (continued)

Variable		Species	Length	Weight	No. of fish	Year
Selenium	r	-0.213082	0.428737	0.347939	0.256749	0.509749
	Prob. > r	0.328971	0.046490	0.112567	0.248735	0.012965
	n	23	22	22	22	23
Zinc	r	-0.880156	0.216331	0.119587	0.250775	0.538405
	Prob. > r	0.008944	0.680566	0.821475	0.631724	0.212476
	n	7	6	6	6	7

The physical variables included in Table 81 include the species of fish, the length and weight of the fish in the sample, and the number of fish that were composited. This latter is somewhat related to the size variables in that more fish tend to be composited when the fish are smaller. The species breakdown is whether the sample was composed of carp (species = 1) or not (other species = 2). As was noted above, a significant portion of the inorganic contaminants samples were from carp (81%); data presented by Wilson *et al.* (1993) indicate that carp are a minor component of the diet of nesting least terns. These variables along with the sample type are also evaluated by correlation analysis with each other (Table 81). Sample type was coded as whole fish = 1 and fillet = 2; it correlates positively with species, fish length, and year, the last of which will be discussed separately below. This indicates that in general that fillets came from species other than carp and from larger fish.

Chromium and mercury are positively correlated and zinc is negatively correlated with species (Table 81). This indicates that carp are generally lower in both mercury and chromium, but higher in zinc, than the other species.

Length, in addition to sample type, is positively correlated with weight, mercury, and selenium. This indicates that larger fish have generally higher concentrations of mercury and selenium than smaller fish. An increase in the mercury concentration as fish age is commonly observed (Sorensen, 1991). From the perspective of nesting terns, this would indicate a somewhat reduced threat from inorganic contaminants than would be indicated by the above analysis of body burdens alone. Weight correlates only with length and mercury and does not add any additional insight over that gained from the length correlations.

The number of fish per sample does not correlate with any of the variables. This indicates that the size of the composite is not a factor that affects the results.

Year was included in the correlation analysis to enable a simple trend analysis. The sample set extends over a 14-year inclusive period from 1983 through 1997. A significant correlation with year would indicate either an increasing or decreasing trend in the concentration of the contaminant or in the case of sample, length, weight, and number of fish, the samples themselves. All of these latter variables are correlated with year, indicating that other species than carp have gained in importance in later years, larger fish have become more prevalent in the samples as time goes on, and the samples are composed of fewer fish in the more recent samples.

Cadmium, mercury, and selenium are positively correlated with year (Table 81). This is particularly important in that any threat posed by these contaminants would be increasing over time. This could mean that such a threat would be increasingly important in the future. The opposite would be true of chromium and barium, although there is no indication of any threat from either of these trace elements.

The correlation analysis indicated that there were trends associated with the composition and type of the fish contaminant samples collected. This is further evaluated in Table 82 as it relates to mercury and selenium.

Table 82. Comparison of the distribution of mercury and selenium between fish species and sample types

A. Mercury in fish

Analysis of Variance			Species - Mean (ppm)		Sample type - Mean	
Source	F-ratio	Prob. > F	Carp	Others	Whole fish	Fillet
Species	5.224	0.026	0.114	0.166	0.087	0.194
Sample type	22.494	< 0.001				
Sp. X sample	15.734	< 0.001				

B. Selenium in fish

Analysis of Variance			Species - Mean (ppm)		Sample type - Mean	
Source	F-ratio	Prob. > F	Carp	Others	Whole fish	Fillet
Species	0.599	0.449	1.543	1.026	1.285	1.285
Sample type	0.000	1.000				
Sp. X sample	0.052	0.822				

Table 82 indicates that the mercury concentration varies between species. Species other than carp are generally higher in mercury than carp (Table 82A) on the average. The results also indicate that fillets tend to be higher in mercury than whole fish samples. However, there is also a significant interaction effect between species and sample type (Sp. X sample in Table 82). This would indicate that the any relationship between the mercury and species changes with sample type; that effect may warrant further investigation.

The distribution of mercury in fish tissues is related to the form of the mercury ingested and the time since ingestion (Sorensen, 1991), with methyl mercury (CH_3Hg^{+1}) predominanting in the skeletal muscle with increasing time from ingestion. The fish samples are collected any time between May and September (NDEQ, 1996). Whether this has an effect on the results is unknown. Most of the NDEQ data only include the year, not the date sampled.

Unlike mercury, there is no significant difference in selenium between species or sample type. This would indicate that any conclusions can be taken at face-value.

The fish samples include those from the North Platte, South Platte, and Platte mainstem rivers in Nebraska. The data were also evaluated for differences in mercury and selenium in the 3 rivers based on Fisher's Least Significant Difference (LSD) Test (Table 83). There were not enough observations for the other trace elements to make a statistical comparison. The results of this analysis could indicate a general source of the 2 elements. For example, the results shown in Table 83 indicate that fish from the North and South Platte rivers are significantly higher in mercury than those from the Platte and the fish from the South Platte are significantly higher in selenium than those from either the North Platte or the Platte mainstem. This may indicate that the source of selenium is in the South Platte basin. Alternatively, recent research has indicated that the predominant source of mercury in fish is atmospheric (Porcella, 1994). If the differences among the 3 basins reflect proximity to the source, then it would be to the west.

Table 83. Comparison of mercury and selenium concentrations in fish from the North Platte, South Platte, and Platte rivers

Mercury (ppm)		Fisher's LSD		Probability	
Site	LS Mean	N. Platte	Platte	N. Platte	Platte
N. Platte	0.135				
Platte	0.077	-0.0579		0.0168	
S. Platte	0.159	0.0233	0.0812	0.3364	0.0007
Selenium (ppm)		**Fisher's LSD**		**Probability**	
Site	LS Mean	N. Platte	Platte	N. Platte	Platte
N. Platte	1.022				
Platte	1.183	0.1610		0.7383	
S. Platte	2.160	1.1379	0.9769	0.0265	0.0460

Organic Contaminants

A summary of the organic contaminants data is presented in Table 84. As is shown by the length and weight data, there are 82 samples; however, the most analyses for any individual contaminant is 66 for DDE, followed by 44 analyses for dieldrin and 42 for each of 2 PCB isomers. The emphasis reflects the NDEQ (1996) statement that dieldrin and PCB's were the toxic pollutants of greatest concern in the State, based on fish tissue data. The majority of fish consumption advisories in Nebraska are for excessive PCB's. However, the fish consumption advisories are based on a $1*10^{-5}$ cancer risk in humans from 2 consecutive samples based on EPA's Risk Assessment Method (NDEQ, 1996). The fish consumption advisories would not reflect the potential for toxicity to birds, except in the unlikely event that cancer risk in birds is comparable to that in people.

NIWQP (1998) only includes information on DDT and its degradation products, DDD and DDE. All of the dietary thresholds for DDT, DDD, and DDE were related to eggshell thinning and egg breakage. The NIWQP (1998) thresholds were all well in excess of the maxima shown in Table 84, *i.e.* minimum threshold of 2.5 ppm (DDE) ww, which produced 20 percent eggshell thinning in one experiment and 10 percent egg cracking in another.

Table 84. Summary data for organic contaminants in Platte River basin fish (all in ppm ww, unless otherwise noted)

	LENGTH (in.)	WEIGHT (lb.)	Chlordane	Gamma Chlordane	Pentachloro anisole
Minimum	5.00	0.67	0.00	0.01	0.00007
Median	17.96	2.97	0.09	0.01	0.00037
Maximum	24.80	6.04	0.80	0.01	0.00120
# of Obs.	82	82	17	1	8

	Pentachloro benzene	*cis*-Chlordane	*trans*-Chlordane	*cis*-Nonachlor	*trans*-Nonachlor
Minimum	0.0003	0.0017	0.0018	0.0020	0.002
Median	0.0003	0.0034	0.0035	0.0035	0.004
Maximum	0.0003	0.0370	0.1800	0.1800	0.042
# of Obs.	1	35	27	15	32

	Oxychlordane	Heptachlor	Hexachloro benzene	Heptachlor Epoxide	Aldrin
Minimum	0.0023	0.01	0.0003	0.0014	0.01
Median	0.0050	0.18	0.0011	0.0035	0.01
Maximum	0.0050	0.18	0.0060	0.0120	0.06
# of Obs.	4	4	8	13	4

	Dieldrin	*p,p'*DDT	*p,p'*DDD	*p,p'*DDE	Aroclor®-1248
Minimum	0.002	0.002	0.01	0.003	0.10
Median	0.011	0.009	0.01	0.032	0.10
Maximum	0.180	0.140	0.06	1.380	0.10
# of Obs.	42	14	15	66	1

	PCB-1254	PCB-1242	PCB-1260	Trifluralin	
Minimum	0.02	0.02	0.01	0.01	
Median	0.05	0.09	0.04	0.02	
Maximum	0.71	0.10	0.32	0.10	
# of Obs.	43	4	44	12	

The other contaminants included in Table 84 do not have any dietary thresholds (or NOAEL's) that could be located. This evaluation will not attempt to relate the concentrations of any of the organic contaminants in relation to dietary thresholds. The screening will simply be done by comparing the data to an ambient background for fish. The background being used is simply the geometric mean of the data gathered by the Fish and Wildlife Service's National Pesticide Monitoring Program (NPMB).

The NPMP geometric means and a comparison to the NDEQ data are shown in Table 85. The rationale for the comparison is that if the Platte River data are near the average for the U.S., then about ½ the samples will be greater than the NPMP geometric mean. If the percentage

Table 85. Comparison to geometric wet weight mean concentrations for 1976-81 of the National Pesticide Monitoring Program (Schmitt et al., 1985)

	cis-Chlordane	*trans*-Chlordane	*cis*-Nonachlor	*trans*-Nonachlor	Oxychlor-dane
NPMP Ave. (ppm *ww*)	0.053	0.023	0.020	0.040	0.010
Number > Ave.	0	4	3	2	0
Total Observations	35	27	15	32	4
% > Ave.	0.0	14.8	20	6.3	0

	Heptachlor	Dieldrin	*p,p'*DDT	*p,p'*DDD	*p,p'*DDE
NPMP Ave. (ppm *ww*)	0.013	0.047	0.047	0.083	0.240
Number > Ave.	2	3	1	0	4
Total Observations	4	42	14	15	66
% > Ave.	50	7.1	7.1	0.0	6.1

	Aroclor®-1248	PCB-1254	PCB-1260
NPMP Ave. (ppm *ww*)	0.130	0.393	0.330
Number > Ave.	0	1	0
Total Observations	1	43	44
% > Ave.	0	2.3	0.0

exceeding the NPMP average is much greater than 50 percent, then there would be a potential threat and more in-depth investigation would be recommended.

None of the organic contaminants in the NDEQ data set has 50 percent of the samples greater than the NPMP geometric mean. Two of four heptachlor samples (or 50 percent exactly) were greater than the NPMP benchmark.   Of all of the other organic contaminants, 20 percent or less were above the NPMP benchmark. Based on this result, the organic contaminants in the NDEQ data set are below the average for a normal background data set. This would indicate that organic contaminants are not likely to be anything more than an extremely isolated threat, if any at all, in the Platte River basin.

A correlation analysis was also performed on the organic contaminants data like that for the inorganic data. The results are shown in Table 86. Length only correlates with weight, while weight only correlates with length and percent lipids (fats). This only confirms the obvious, *i.e.* longer fish weigh more and heavier fish contain more fat. However, percent lipids does correlate with heptachlor epoxide (inverse) and two of the DDT congeners, *p,p'*DDT and *p,p'*DDD. Since organochlorine pesticides tend to correlate with fat content (NIWQP, 1998), this should come as no surprise either.   However, they also indicate that DDT correlates with fish age and length. The NDEQ samples do not show the latter correlation, indicating that accumulation is not occurring in the Plate basin.

Table 86. Correlations among concentrations of organic contaminants, physical variables, and

		Length	Weight	% Lipids	Year
Length	r	—	0.840094	0.349912	0.494449
	Prob. > r	—	< 0.000001	0.183987	0.000002
	n	82	82	16	82
Weight	r	0.840094	—	0.645199	0.417626
	Prob. > r	< 0.000001	—	0.006955	0.000095
	n	82	82	16	82
Chlordane	r	-0.044601	-0.113274	-0.287316	-0.460820
	Prob. > r	0.865033	0.665113	0.391620	0.062655
	n	17	17	11	17
Chloroanisol	r	-0.122774	0.023134	No Data	-0.556437
	Prob. > r	0.772101	0.956640	—	0.152033
	n	8	8	0	8
cis-Chlordane	r	-0.141857	-0.309192	-0.294073	-0.444146
	Prob. > r	0.416287	0.070708	0.329434	0.007521
	n	35	35	13	35
trans-Chlordane	r	-0.098599	-0.202351	0.099744	-0.340151
	Prob. > r	0.631796	0.321514	0.770450	0.089082
	n	26	26	11	26
cis-Nonachlor	r	-0.114737	-0.307105	-0.258810	-0.348640
	Prob. > r	0.696112	0.285506	0.575196	0.221835
	n	14	14	7	14
trans-Nonachlor	r	0.005054	-0.183411	-0.141952	-0.361694
	Prob. > r	0.978472	0.323331	0.643652	0.045572
	n	31	31	13	31
Hexachlorobenzene	r	0.006532	0.025634	No Data	-0.877330
	Prob. > r	0.987752	0.951958	—	0.004201
	n	8	8	0	8
Heptachlor Epoxide	r	-0.435346	-0.490074	-0.970851	-0.597403
	Prob. > r	0.157209	0.105789	0.029149	0.040246
	n	12	12	4	12
Dieldrin	r	-0.046809	-0.140415	-0.28197	-0.115473
	Prob. > r	0.774256	0.387481	0.498661	0.477994
	n	40	40	8	40
*p,p'*DDT	r	0.047906	-0.004572	0.999998	-0.253146
	Prob. > r	0.882467	0.988748	0.001337	0.427275
	n	12	12	3	12
*p,p'*DDD	r	-0.055715	-0.084603	0.794399	0.050951
	Prob. > r	0.849955	0.773687	0.010543	0.862668
	n	14	14	9	14

Table 86. (continued)

		Length	Weight	% Lipids	Year
*p,p'*DDE	r	0.034522	0.074779	0.074010	-0.099661
	Prob. > r	0.786536	0.557024	0.785315	0.425936
	n	64	64	16	66
PCB-1254	r	-0.277396	-0.288737	-0.824650	-0.489560
	Prob. > r	0.075311	0.063672	0.382745	0.001000
	n	42	42	3	42
PCB-1260	r	0.017286	-0.026784	-0.829863	-0.188009
	Prob. > r	0.911323	0.862979	0.082059	0.221655
	n	44	44	5	44
Trifluralin	r	0.175660	-0.159522	No Data	-0.404289
	Prob. > r	0.585010	0.620444	—	0.192405
	n	12	12	0	12
% Lipids	r	0.349912	0.645199	—	-0.201575
	Prob. > r	0.183987	0.006955	—	0.454078
	n	16	16	16	16

The 4th set of correlations in Table 86 is a trend analysis based on year. As was the case with the inorganic contaminants, there are significant positive correlations between length and weight and year. It should be noted that the fish in the inorganic contaminants data set are not necessarily the same as those in the organic data set. There are 82 observations in the organic data sets for length and weight (Table 86), while there are only 62 in the inorganic data set for length and weight (Table 81). The results are still the same, *i.e.*, r-values of around 0.4 and an indication that the size of the fish sampled has increased over time.

There are 5 significant correlations between organic contaminant concentrations in fish and year (Table 86). All 5 are negative. As a matter of fact, when the nonsignificant correlations between year and organic contaminant concentration are considered, only one is positive, the near-zero correlation with *p,p'*DDD. In other words the concentrations of organic contaminants have been decreasing over time, some significantly, some not. This is no surprise; the use of nearly all organochlorine pesticides has been banned in the U.S. since about the beginning of the monitoring period. Furthermore the use of the non-pesticide organochlorines in the data set, *i.e.*, PCB's, has also been phased out over the years.

The only non-organochlorine included in Table 86 - trifluralin, a trifluoro, dinitro-aniline herbicide, is considered of low toxicity to birds. The dietary LC_{50} is >5000 ppm in both quail and mallard (Ahrens, 1994). Trifluralin also has a negative r-value with year, although it is not statistically significant.

Trends

Fish samples for inorganic and organic contaminants have been collected since 1983 and analyzed for a variety of inorganic and organic contaminants by the NDEQ. Data for the years 1983 through 1997 have been provided to Reclamation for this evaluation. It should be noted that the samples have been collected for a completely different purpose than this application. This does affect the results somewhat.

Over the years there has been a trend toward an increasing percentage of fillet samples as opposed to whole fish. The latter are more appropriate for an evaluation of the potential effects on predatory wildlife. There has also been a trend toward increasingly larger fish in terms of both length and weight. The focus of this evaluation is related to food items for the least tern, which takes only very small fish as food. These trends have the potential to confound results related to trends in contaminants in that their concentrations are often related to the size of the fish and the tissue being sampled.

The samples consist of composites of between 1 and 11 fish. There has been a significant trend toward a decreasing number of fish per composite sample. This is consistent with an increase in the size of the fish being sampled. This in and of itself should not affect the results.

Among the inorganic analytes, there has been an increasing trend in cadmium, mercury, and selenium (Table 81). Both mercury and selenium are correlated with fish size. Consequently the trend may be an artifact of the trend toward larger fish in the samples. The data do not allow for an evaluation of this.

There was a decreasing trend in both barium and chromium. Neither of these trace elements are of particular concern. The decreasing trend would further reduce any concern that did exist.

Five of the organic contaminants showed significant trends over time. All were decreasing trends and included *cis*-chlordane, *trans*-nonachlor, hexachlorobenzene, heptachlor epoxide, and PCB-1254. Actually all but one of the nonsignificant trends are also negative. Since most of the organics are chlorinated hydrocarbons that have been banned for further use in the United States, the decline should be expected.

Recommendations for future monitoring

Seven of the 10 inorganic analytes were compared to levels of concern (see Table 81). Of the 7, 5 exceeded a presumed no effect level. These included cadmium, lead, mercury, selenium, and zinc. There were fewer than 10 samples of lead and zinc, but over 50 percent of the samples that were available exceeded their respective level of concern. Because of the high percentage and the paucity of samples, additional data on both lead and zinc appear necessary. Lead, along with cadmium and mercury are known as the "big three" among heavy metal poisons (Manahan, 1989). Cadmium and mercury are also elevated in 89 and 44 percent of the samples respectively. There are more analyses for mercury than for any other element. The remaining element that exceeded its level of concern is selenium, which was the only element to also exceed its toxicity threshold. Selenium exceeded the level of concern in 78 percent of the samples. All 5 of these elements are recommended for future monitoring.

The samples analyzed to date by the NDEQ are made up of larger fish. As was noted above, all of the samples consisted of fish that are much larger than those suitable as prey for least terns. Consequently any future monitoring should focus on smaller fish, *i.e.* < 1.5 inches. In addition, there are no samples of the type of food items taken by piping plovers. This gap in data could be filled by also collected samples of aquatic invertebrates for contaminants analysis. Analytes of concern would at a minimum consist of the 5 elements listed above. However, since there are no data available on the contaminants in aquatic invertebrates, a wider suite of analytes could be considered.

All of the organic analytes were present in concentrations that appear to be consistent with an uncontaminated background. Because all are showing decreasing trends over time as well, none of these appear to constitute a threat to breeding birds.

Atrazine has exceeded its water quality standard in a significant percentage of the samples that have been collected in recent years. For this reason it is considered a potential contaminant in fish. There are no data on atrazine concentrations in fish. Consequently it is recommended that future monitoring for contaminants in fish also include atrazine among the analytes.

Biological Contaminants in Bird Eggs from the Platte River

The Fish and Wildlife Service sampled bird eggs from the Platte River near Grand Island during 1991 through 1993, with one additional sample collected during 1994. A breakdown of the samples by species and by year is included in Table 87. In addition to the samples summarized in Table 87, there were 2 killdeer eggs and 4 unidentified eggs collected in 1991, 2 American kestrel eggs collected in 1992, and 3 peregrine falcon eggs collected in 1993. This report will only address the analyses of the least tern and piping plover eggs.

Table 87. Summary of Egg Samples Collected from the Platte River

Species	1991	1992	1993	1994	Total
Least Tern	10	21	6	1	38
Piping Plover	2	23	39	0	64
Total	12	44	45	1	108

The samples collected during 1991 were analyzed for 19 elements, including aluminum, arsenic, barium, beryllium, boron, cadmium, chromium, copper, iron, lead, magnesium, manganese, mercury, molybdenum, nickel, selenium, strontium, vanadium, and zinc. During 1992, ½ the samples were analyzed for the full suite of analytes, while the other ½ were only analyzed for arsenic, mercury, and selenium. The 1993 samples were only analyzed for the latter 3 elements. The 1994 sample was once again analyzed for the full suite of elements. In addition to the elements listed above, sample weight and percent moisture were also reported for most of the samples, *i.e.*, 95 of 102.

The least tern and piping plover egg data are summarized in Table 88. Part A of the table summarizes the least tern data and Part B summarized the piping plover data. The data are presented separately because there are many interspecific differences in the measured variables. The differences are evaluated in Table 89, based on Mann-Whitney tests. The Mann-Whitney test is a nonparametric equivalent of a simple t-test.

Mercury shows the most significant difference between the species. However, from the data in Table 89, the species with the higher value cannot be readily determined. The data in Table 88 do not readily show the difference. For example, the minimum mercury concentrations are 0.23 and 0.08 ppm in the least tern and piping plover egg data respectively. The maximum mercury concentrations in the 2 sets of egg data are nearly equal at 4.4 and 4.6 ppm. However, the median mercury concentrations are very different and reflect the distributions of the data being evaluated, and they show the difference very well. A plot of the data will better illustrate the difference. As can be seen from Figure 177, there is very little overlap in the 2 data sets. The minima and maxima are about the only points at which there is any near-coincidence of the data. Between those 2 points, the least tern distribution shows a somewhat linear increase, while the piping plover distribution is extremely concave. Consequently, the medians diverge considerably and reflect the statistical difference.

Table 88. Summary of Platte River Endangered Species Egg Contaminants Data - all in mg/Kg dry weight (or ppm) unless otherwise noted)

A. Least Tern

	Weight (gm)	Moisture (%)	Aluminum	Arsenic	Boron	Barium	Beryllium
N of cases	37	37	19	36	19	16	18
Minimum	1.73	2.68	3.0	0.1	0.5	0.68	0.01
Maximum	17.96	79.80	91.9	0.3	7.0	11.00	1.00
Median	5.10	74.90	19.6	0.2	2.0	3.55	0.06
No. < D.L.	-----	-----	7	34	14	6	17

	Cadmium	Chromium	Copper	Iron	Mercury	Magnesium	Manganese
N of cases	19	19	19	19	30	19	19
Minimum	0.01	0.10	2.00	70.7	0.23	254	1.06
Maximum	0.88	1.99	4.37	206.0	4.40	832	116.00
Median	0.10	0.50	3.60	128.0	1.55	411	4.19
No. < D.L.	18	14	0	0	0	0	0

	Molybdenum	Nickel	Lead	Selenium	Strontium	Vanadium	Zinc
N of cases	19	19	19	38	19	19	19
Minimum	0.5	0.05	0.034	1.10	1.50	0.3	16.3
Maximum	4.0	4.49	1.100	7.79	28.00	1.0	82.1
Median	0.8	0.50	0.300	4.90	6.94	0.5	60.6
No. < D.L.	18	16	17	0	0	18	0

Table 88 (continued)
B. Piping Plover

	Weight (gm)	Moisture (%)	Aluminum	Arsenic	Boron	Barium	Beryllium
N of cases	56	56	15	61	15	14	14
Minimum	1.48	52.30	3	0.20	0.5	2.7	0.02
Maximum	7.70	76.10	55	0.67	10.0	15.2	0.10
Median	5.00	71.30	6	0.20	2.0	6.8	0.02
No. < D.L.	-----	-----	9	57	11	1	14

	Cadmium	Chromium	Copper	Iron	Mercury	Magnesium	Manganese
N of cases	14	14	14	14	59	14	14
Minimum	0.04	0.1	2.6	68.0	0.08	311.0	1.0
Maximum	0.10	0.5	4.2	172.0	4.60	588.0	3.8
Median	0.04	0.1	3.3	86.5	0.25	392.5	1.8
No. < D.L.	14	11	0	0	0	0	0

	Molybdenum	Nickel	Lead	Selenium	Strontium	Vanadium	Zinc
N of cases	14	14	14	60	13	14	14
Minimum	0.5	0.1	0.3	2.7	4.3	0.3	33.0
Maximum	2.0	0.5	2.0	15.0	31.0	0.7	85.7
Median	1.0	0.1	0.6	5.3	12.3	0.3	51.6
No. < D.L.	14	11	14	0	0	14	0

Table 89. Comparison of Egg Data by Species based on the Mann-Whitney U Test

Variable	Tern n	Plover n	X^2	U	Probability
Weight	36	56	0.0270	1028.5	0.869380
Moisture (%)	36	56	10.4243	1411.5	0.001244 **
Aluminum	18	15	1.5696	169.0	0.210265
Arsenic	35	61	0.2753	1015.5	0.599799
Boron	18	15	1.6846	100.0	0.194309
Barium	15	14	8.0391	40.5	0.004578 **
Beryllium	17	14	5.7912	170.0	0.016107 *
Cadmium	18	14	5.6619	179.0	0.017338 *
Chromium	18	14	4.5421	178.0	0.033072 *
Copper	18	14	5.5663	188.0	0.018310 *
Iron	18	14	9.3613	206.5	0.002216 **
Mercury	29	59	44.7102	1608.5	< 0.000001 ***
Magnesium	18	14	1.2987	156.0	0.254451
Manganese	18	14	10.1968	210.0	0.001407 **
Molybdenum	18	14	3.6405	77.5	0.056389
Nickel	18	14	4.5821	178.0	0.032307 *
Lead	18	14	5.4784	73.5	0.019253 *
Selenium	37	60	1.1764	964.0	0.278100
Strontium	18	13	4.9393	61.5	0.026253 *
Vanadium	18	14	1.6006	157.0	0.205820
Zinc	18	14	1.3426	156.5	0.246576

Asterisks flag statistically significant tests as follows: * probability of greater $X^2 < 0.05$; ** prob. < 0.01; *** prob. < 0.001. Of the 21variables tested, 12 show statistically significant differences.

There are also highly significant differences (probability < 0.01) in the concentrations of several of the variables, including percent moisture, barium, iron, and manganese. It should be noted that the minimum moisture content in the tern eggs was less than 3 percent; this would represent a desiccated egg that would probably not constitute a representative sample. However, it was not discarded. One outlier would not be a significant factor in the nonparametric analysis, wherein outliers have no more influence than any other single observation.

There are also 7 elements that show significant differences in Table 89 (probability < 0.05). These include beryllium, cadmium, chromium, copper, nickel, lead, and strontium. The results for the majority of these elements are below the detection limits (Table 88). The exceptions are copper and strontium. The reason for the differences between species relates to the variation in detection limits between years that coincides with a variation in sample species composition between years as well. These differences are more a difference due to sampling bias than a real difference between species.

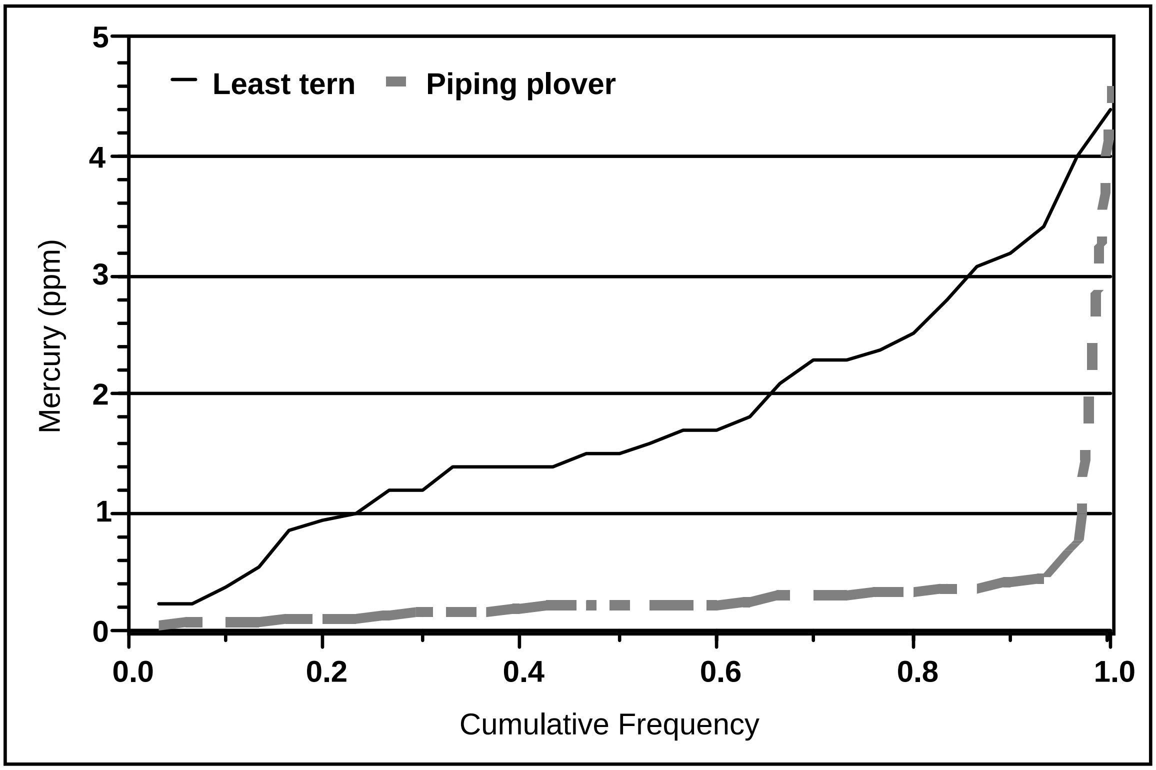


Figure 177. Cumulative frequency distributions of mercury in the least tern and piping plover egg data sets from 1991 through 1994

The effect of the change in detection limit on the results can be illustrated using the cadmium data. Only one sample had a measurable cadmium concentration (Table 88), a least tern collected in 1991. The complete cadmium data set for 1991 and 1992 is shown on Figure 178. The plot includes the data for other species (unidentified, kestrels, and killdeer) collected in the 2 years. There was no measurable cadmium in any of those samples.

The majority of samples collected in 1991 were tern eggs with a detection limit of 0.1 ppm (Figure 178). The majority of samples collected in 1992 were piping plover eggs with a detection limit of 0.04 ppm. This dichotomy of sampled species and detection limits appears to cause the difference in the cadmium concentration between the tern and plover eggs. It is also the primary control on the other trace elements, including (with their respective 1991 and 1992 detection limits): barium (5 and 0.1-0.4 ppm), beryllium (0.1 and 0.02 ppm), chromium (0.5 and 0.1-0.4 ppm), nickel (0.5 and 0.1-0.4 ppm), and lead (0.3 and 0.6 ppm). For the trace elements that are mostly below detection limits, there is little concern over potential toxicity.

The detection limits also varied between years for most of the other elements, but as long as the results were in the quantifiable range, this has no effect on the results. These are the elements that can be evaluated for potential toxicity.

The elements for which there are NIWQP (1998) guidelines that relate to the Platte River egg samples are compared in Table 90. Of the elements shown in Table 90, a very high percentage

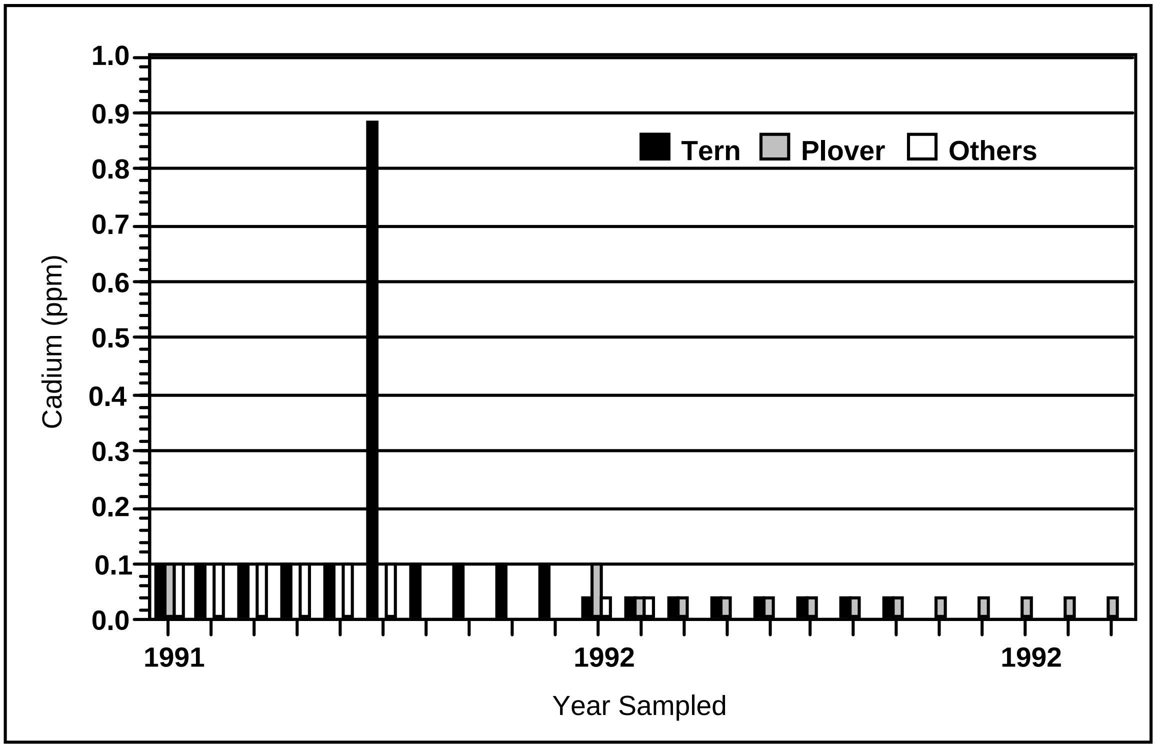


Figure 178. Cadmium concentrations in all samples collected in 1991 and 1992

of the arsenic (94%), boron (74%), and molybdenum (97%) are below their respective laboratory reporting limits. None of these elements exceeds its NOAEL (Table 90).

Copper was present in measurable quantities in all of the eggs (Table 88). All of the least tern and plover eggs were below the NOAEL (Table 90). However, one of the killdeer eggs had a copper concentration of 5.53 ppm, which is in reality approximately equal to the NOAEL. However, the eggs of the other species as a group do not differ significantly in their copper content from either the tern or the plover eggs ($X^2 = 0.85$ and 2.26 respectively for the Mann-Whitney tests). Copper does not appear to be a major threat to the reproduction of endangered birds based on these egg data.

A very high percentage of the least tern eggs had a mercury concentration in excess of the NOAEL. A relatively small percentage of the plover eggs had mercury levels that were above the NOAEL. This reflects the above mentioned interspecific difference in the mercury concentration in the eggs. It should be noted that the NOAEL used for comparison may be extremely conservative. For example, Thompson (1996) recommended a no effect threshold of 2.5 ppm dw. About 23 percent of the least tern eggs exceeded that level, while less than 2 percent of the plover eggs were greater than Thompson's no effect threshold.

The NIWQP (1998) recommended a toxicity threshold for mercury of 3.4 ppm in bird eggs. About 7 percent of the tern eggs exceeded the toxicity threshold. The same plover eggs that exceeded Thompson's no effect threshold also exceeded the NIWQP (1998) toxicity threshold.

Table 90. Comparison of contaminants data to No Observed Adverse Effects Level [NOAEL] taken from NIWQP (1998)

Element	NOAEL [ppm dw]	Species	Year	% > NOAEL
Arsenic	1.3	Least Tern	1991-93	0.0%
		Piping Plover	1991-93	0.0%
Boron	52	Least Tern	1991-92	0.0%
		Piping Plover	1991-92	0.0%
Copper	5.5	Least Tern	1991-92	0.0%
		Piping Plover	1991-92	0.0%
Mercury	0.4	Least Tern	1991-93	90.0%
		Piping Plover	1992-93	13.1%
Molybdenum	13	Least Tern	1991-92	0.0%
		Piping Plover	1991-92	0.0%
Selenium	3.0	Least Tern	1991-93	86.8%
		Piping Plover	1991-93	93.5%
Zinc	50	Least Tern	1991-92	63.2%
		Piping Plover	1991-92	50.0%
Cadmium -	Furness (1996) indicates that Cd is almost always low in bird eggs (usually < D.L. by AA)			
Lead - - - -	Pain (1996) indicates that blood, liver, and kidney concentrations are primary indicators of contamination			

The intermediate zone between the NOAEL and the toxicity threshold is labeled the level of concern by NIWQP (1998). This is a level at which some effects on embryonic birds could be expected, particularly to sensitive individuals. The vast majority of the tern eggs fall within the level of concern. Consequently mercury should be labeled a concern for breeding birds in the study area.

A majority of both the least tern and plover eggs exceeded the NIWQP (1998) NOAEL for selenium (Table 90). Unlike that for mercury, a greater percentage of the plover eggs exceeded the NOAEL than did the tern eggs, although in this case the difference is small. There was no interspecific difference in the selenium content of the eggs. Like the NIWQP (1998) NOAEL for mercury, the one for selenium may also be very conservative. Heinz (1996) recommended a no effect level for selenium in eggs of 3 ppm ww (or approximately 12 ppm dw assuming a moisture content of 75 percent). This latter is greater than the toxicity threshold for selenium recommended by the NIWQP of 6 ppm dw. The difference in the various recommendations may reflect the basis for the recommendations. Heinz (1996) recommendation is based on controlled feeding studies, while the NIWQP (1998) toxicity threshold is based on field data. In the case of field studies, the toxicity may reflect either the synergistic effect of other contaminants or the effect of some other contaminant. Alternatively field studies do reflect the real world where confounding effects would be the rule rather than the exception.

The comparison of the egg data to the NIWQP (1998) selenium toxicity threshold showed that none of the tern eggs exceeded it, but about 29 percent of the plover eggs did. In either case the majority of the eggs were within the level of concern. Consequently selenium would have to be added to the list of contaminants of concern for both least tern and piping plover reproduction in the Platte River.

A significant percentage of both tern and plover eggs exceeded the NOAEL of zinc (Table 90). NIWQP (1998) did not recommend a toxicity threshold for zinc. The rationale for this was that zinc toxicity thresholds are not well established because zinc concentrations are homeostatically regulated (NIWQP, 1998). Nevertheless no matter what such a threshold might be, under the definition of the "level of concern" the zinc data would fall within it. On the basis of this rationale, zinc should also be added to the list of contaminants of concern.

Trends

There are only 2 years of available data for most constituents, and 3 at most for several others. This is an insufficient data base for a trend analysis.

Recommendations for future monitoring

Eggs of terns and plovers were collected during the years 1991-93. In 1991 and 1992 (about ½ the samples in 1992), the eggs were analyzed for their content of aluminum, arsenic, barium, beryllium, boron, cadmium, chromium, copper, iron, lead, magnesium, manganese, mercury, molybdenum, nickel, selenium, strontium, vanadium, and zinc. The remaining eggs in 1992 and those collected in 1993 were analyzed for arsenic, mercury, and selenium only.

Mercury, selenium, and zinc were elevated in a large percentage of the eggs. A small percentage of the eggs exceeded the toxicity threshold for mercury. There are no data on the state of the eggs or the embryos within; so actual effects cannot be confirmed. However, samples that are in excess of the no effect level may or may not exhibit adverse effects. Since one of the eggs was extremely desiccated (% moisture < 3), some cause of mortality was present. The desiccated egg did exceed the no effect level for selenium, although not the toxicity threshold. Nevertheless all 3 elements are within the level of concern for potential embryonic toxicity and should be monitored in the future.

References

Admiraal, Davis, John Stansbury, John Holz, Kyle Hoagland, and Cory Haberman. 2003. A Physical Model of Flow Patterns in Lake Ogallala. A Study Conducted for the Nebraska Game and Parks Commission. University of Nebraska, Lincoln, Nebraska. 69 pp. + App.

Ahrens, William H. (Ed.). 1994. Herbicide Handbook, Seventh Edition - 1994. Weed Society of America, Champaign, Illinois. 352 pp.

Anderson, M.S., H.W. Lakin, K.C. Besson, Floyd F. Smith, and Edward Thacker. 1961. Selenium in Agriculture. Agriculture Handbook No. 200, U.S. Department of Agriculture, Agricultural Research Service, Washington, D.C. 64 pp.

Ayers, R.S. and D.W. Westcot. 1985. Water Quality for Agriculture. FAO Irrigation and Drainage Paper 29, Rev. 1. Food and Agriculture Organization of the United Nations, Rome, Italy. 174 pp.

Babcock, H.M., and F.N. Visher. 1951. Ground-Water Conditions in the Dutch flats Area, Scotts Bluff and Sioux Counties, Nebraska, with a Section on the Chemical Quality of the Ground Water by W.H. Durum. Geological Survey Circular 126, USGS, Washington, D.C. 39 pp. + 2 plates in pocket.

Bowman, Gilbert A., Larry G. Ragon, Charles L. Hammond, Lawrence E. Brown, and Raphael A. Boccheciamp. 1971. Soil Survey of Phelps County, Nebraska. U.S. Department of Agriculture, Soil Conservation Service (now, Natural Resources Conservation Service) and University of Nebraska, Conservation and Survey Division, Lincoln, Nebraska. 91 pp. + 46 maps + 2 Indices.

Bureau of Reclamation. 2006. Report of Actual Operations – North Platte Basin. Downloaded on Feb. 10, 2006, from:
http://www.usbr.gov/dataweb/html/northplatte.html

Bureau of Reclamation. 2005. Report of Actual Operations – North Platte Basin. Downloaded on Feb. 17, 2006, from:
http://www.usbr.gov/gp/aop/np/0405/2004actual.cfm

Burns, Alan. 1983. Hydrologic Data from the Tamarack Wildlife Area and Vicinity, Logan County, Colorado. U.S. Geological Survey Open-File Report 83-139, USGS, Lakewood, Colorado. 123 pp. + 2 plates in pocket.

Burns, Alan. 1985. Hydrologic Description of the Tamarack Wildlife Area and Vicinity, Logan County, Colorado, and Simulated Effects of Possible Water-Management Activities. U.S. Geological Survey Water-Resources Investigations Report 85-4014, USGS, Lakewood, Colorado. 42 pp. + 1 plate in pocket.

Cannon, Helen L. 1960. The Development of Botanical Methods of Prospecting for Uranium on the Colorado Plateau. Geological Survey Bulletin 1085-A, United States Government Printing Office, Washington, D.C. 50 pp. + 2 Plates in Pocket.

Church, S.E., B.A. Kimball, D.L. Fry, D.A. Federer, T.J. Yager, and R.B. Vaughn. Source, Transport, and Partitioning of Metals between Water, Colloids, and Bed Sediments of the Animas river, Colorado. U.S. Geological survey Open-File Report 97151, USGS, Denver, Colorado. 135 pp.

Cole, Gerald A. 1979. Textbook of Limnology. 2nd Ed. The C.V. Mosby Company, St. Louis, Missouri. 426 pp.

Colorado Department of Health and Environment. 1999. Status of Water Quality in Colorado 1998. Corrected in January 1999. CDPHE, Denver, Colorado. Numbered by Chapter.

Corbella, Jacinto, and Jose L. Domingo. 1996. Developmental Effects of Aluminum, Uranium, and Vanadium. In: Toxicology of Metals, Louis W. Chang (Ed.), Lewis Publishers, Boca Raton, Florida. Pp: 1083-95.

Crist, Marvin A. 1974. Selenium in Waters in and Adjacent to the Kendrick Project, Natrona County, Wyoming. Geological Survey Professional Paper 2023, United States Government Printing Office, Washington, D.C. 39 pp. + 3 Plates in Pocket

Deason, Jonathan P. 1986. U.S. Department of the Interior Investigations of Irrigation-Induced Contamination Problems. In: Toxic Substances in Agricultural Water Supply and Drainage, Defining the Problems. Proceedings of the 1986 Regional Meetings, U.S. Committee on Irrigation and Drainage. USCID, Denver, Colorado. Pp: 201-210.

Dinan, Kenneth F. 1992. Application of the Stream Network Temperature Model (SNTEMP) to the Central Platte River, Nebraska. Professional Paper, Colorado State University, Ft. Collins, Colorado. 87 pp.

Dove, Eric, John Stansbury, David Admiraal, Kyle Hoagland, and John Holz. 2002. Water Quality Modeling of Lake Ogallala Using Ce-Qual-W2 Model. A Study Conducted for the Nebraska Game and Parks Commission. University of Nebraska, Lincoln, Nebraska. 79 pp.

Druliner, A.D., Brent J. Esmoil, and J. Mark Spears. 1999. Field Screening of Water Quality, Bottom Sediment, and Biota Associated with Irrigation Drainage in the North Platte Project Area, Nebraska and Wyoming, 1995. U.S. Geological Survey Water- Resources Investigations Report 98-4210, USGS, Lincoln, Nebraska. 43 pp.

Druliner, A.D., and T.S. McGrath. 1996. Predicting Nitrate-Nitrogen and Atrazine Contamination in the High Plains Aquifer in Nebraska. U.S. Geological Survey Water-Resources Investigations Report 95S4202, USGS, Lincoln, Nebraska.

Engberg, R.A. and R.F. Spalding. 1978. Groundwater Quality Atlas of Nebraska. Resource Atlas No. 3/1978. Conservation and Survey Division, Institute of Agriculture and Natural Resources, The University of Nebraska–Lincoln, Lincoln, Nebraska. 39 pp.

Facciani, Stephen, and George T. Baxter. 1977. Commercial Fisheries Investigations for Wyoming Reservoirs 1974-1976. Wyoming Game and Fish Department, Cheyenne, Wyoming. 34 pp

Fortescue, John A.C., 1992. Landscape Geochemistry: Retrospect and Prospect – 1990. Applied Geochemistry 7:1-51.

Frenzel, S.A., R.B. Swanson, T.L. Huntzinger, J.K. Stameer, P.J. Emmons, and R.B. Zelt. 1998. Water Quality in the Central Platte Basins, Nebraska, 1992-95. U.S. Geological Survey Circular 1163, USGS, Lincoln, Nebraska. 33 pp.

Furness, Robert W. 1996. Cadmium in Birds. In: Environmental Contaminants in Wildlife - Interpreting Tissue Concentrations. W. Nelson Beyer, Gary H. Heinz, and Amy W Redmon-Norwood (Eds.), SETAC Special Publication Series, Lewis Publishers, Boca Raton, Florida. Pp. 389-404.

Gosselin, David C., Jacqueline Headrick, Xun-Hong Chen, Scott Summerside, Rod Tremblay, and Kurt Bottger. 1996. Domestic Well-water Quality in Rural Nebraska. Data-Analysis Report, Conservation and Survey Division, Institute of Agriculture and Natural Resources, University of Nebraska-Lincoln and Nebraska Department of Health, Lincoln, Nebraska. Numbered by Section.

Grabowski, Stephen J., and James J. Sartoris. 1984. North Platte Hydroelectric Study Fishery and Limnological Investigations 1982–1983. Unpublished report to the LM Region, Engineering and Research Center, Bureau of Reclamation, Denver, Colorado.

Greenberg, Arnold E., Lenore S. Cleseri, and Andrew D. Eaton (eds.). 1992. Standard Methods for the Examination of Water and Wastewater. 18th Edition. American Public Health Association, American Water Works Association, and Water Environment Federation, Washington, D.C. Numbered by Chapter.

Grier, Bob. 2005. Lake Minatare: A Surprising Discovery. NEBRASKAland Magazine, Nebraska Game and Parks Commission, Lincoln, Nebraska. Downloaded on September 12, 2005 from: http://www.ngpc.state.ne.us/nebland/articles/fishing/minatare.asp

Heath, Alan G. 1995. Water Pollution and Fish Physiology. 2nd Ed. CRC Press, Inc., Boca Raton, Florida. 359 pp.

Heinz, Gary H. 1996. Selenium in Birds. In: Environmental Contaminants in Wildlife - Interpreting Tissue Concentrations. W. Nelson Beyer, Gary H. Heinz, and Amy W Redmon-Norwood (Eds.), SETAC Special Publication Series, Lewis Publishers, Boca Raton, Florida. Pp. 447-458.

Hem, John D. 1985. Study and Interpretation of the Chemical Characteristics of Natural Waters. United States Geological Survey Water-Supply Paper 2254, U.S. Government Printing Office, Washington, D.C. 263 pp. + 2 plates in pocket.

Hoagland, Kyle D., John C. Holz, Tadd M. Barrow, and Richard J. Ruane. 2000. Oxygen Budget for Lake Ogallala. A Report to the Nebraska Game and Parks Commission. University of Nebraska, Lincoln, Nebraska and Reservoir Environmnetla Management, Inc., Chattanooga, Tennessee. 19 pp. + tables & figures + 15 App.

Irwin, Roy J., Mark Van Mouwerik, Lynette Stevens, Marion Dubler Seese, and Wendy Basham. 1997. Environmental Contaminants Encyclopedia. National Park Service, Water Resources Division, Fort Collins, Colorado. (Distributed within the Federal Government as an Electronic Document.)

Leopold, Luna B. 2000. Temperature profiles and bathymetry of some high mountain lakes. Proc. Nat. Acad. Sci. 97(12):6267-6270.

Lowe, T.P., T.W. May, W.G. Brumbaugh, and D.A. Kane. 1985. National Contaminant Biomonitoring Program: Concentrations of Arsenic, Cadmium, Copper, Lead, Mercury, Selenium, and Zinc in U.S. Freshwater Fish, 1978-81. Arch. Environ. Contam. Toxicol. 14:363-388.

Manahan, Stanley E. 1989. Toxicological Chemistry. Lewis Publishers, Inc. Chelsea, Michigan. 317 pp.

Miller, William J. 1994. A Review of "Application of the SNTEMP Model to the Central Platte River, Nebraska" by Mr. Kenneth F. Dinan. Report to the Central Nebraska Public Power and Irrigation District. W.J. Miller and Associates, Ft. Collins, Colorado. 34 pp.

Miller, William J. 1996. Analyses of the relationship between water temperature and selected meteorologic and hydrologic variables for the central Platte River, Nebraska, summer 1989-95. Report to the Nebraska Public Power District and the Central Nebraska Public Power and Irrigation District, Miller Ecological Consultants, Inc., Fort Collins, Colorado. 52 pp.

Miner, Jeffrey C., and Roy A. Stein. 1996. Detection of Predators and Habitat Choice by Small Bluegills: Effects of Turbidity and Alternative Prey. Trans. Am. Fish. Soc. 125:97-103

National Irrigation Water Quality Program. 1998. Guidelines for the Interpretation of Biological Effects of Selected Constituents in Biota, Water and Sediment. National Irrigation Water Quality Program Information Report No. 3. United States Department of the Interior, NIWQP, Denver, Colorado. 198 pp. + 2 App.

National Oceanic and Atmospheric Administration. 1999. NOAA SQuiRTs™: Screening Quick Reference Tables. Hazmat Report 99-1. Downloaded 10/4/02 from: http://response.restoration.noaa.gov/cpr/sediment/squirt/squirt.pdf

Nebraska DEQ. 1996a. Title 117 - Nebraska Surface Water Quality Standards. Water Quality Division, Department of Environmental Quality, Lincoln, Nebraska. Paged by Chapter.

Nebraska DEQ. 1996b. Title 118 - Nebraska Ground Water Quality Standards and Use Classification. Department of Environmental Quality, Lincoln, Nebraska. 26 pp. + App.

Nebraska DEQ. 1996c. 1996 Nebraska Water Quality Report. Water Quality division, Department of Environmental Quality, Lincoln, Nebraska. 149 pp. + App.

Nebraska DEQ. 1998. 1998 Nebraska Section 303(d) List of Impaired Waters. Water Quality Division, Department of Environmental Quality, Lincoln, Nebraska. Available at: http://www.deq.state.ne.us/SurfaceW.nsf/Pages/Sec_303D

Nebraska Department of Environmental Quality. 2002a. TITLE 117 - NEBRASKA SURFACE WATER QUALITY STANDARDS. NDEQ, Lincoln, Nebraska. Variously numbered.

Nebraska Department of Environmental Quality. 2002b. Executive Summary of Nebraska's 2002 303(d) List. NDEQ, Lincoln, Nebraska. 2 pp.

Nebraska Department of Environmental Quality. 2003a. Total Maximum Daily Loads for the North Platte River – Parameters of Concern: Fecal Coliform Bacteria. NDEQ, Lincoln, Nebraska. 26 pp. + 2 App.

Nebraska Department of Environmental Quality. 2003b. Total Maximum Daily Loads for the Middle Platte River – Parameters of Concern: Fecal Coliform Bacteria. NDEQ, Lincoln, Nebraska. 23 pp. + 2 App.

Nebraska Department of Environmental Quality. 2004a. 2004 Surface Water Quality Integrated Report. DEQ, Water Quality Division, Lincoln, Nebraska. Variously numbered.

Nebraska Department of Environmental Quality. 2004b. 2004 Nebraska Groundwater Quality Monitoring Report. NDEQ, Water Quality Assessment Section, Groundwater Unit, Lincoln, Nebraska. 26 pp. + App.

Nebraska Department of Natural Resources. 2004. Discharge Records of Streams, Canals, Pumps and Storage in Reservoirs. Hydrographic Report, NDNR, Lincoln, Nebraska. 217 pp.

Nebraska Game and Parks Commission. 2005. 2005 Nebraska Fishing Guide Fishing Regulations and Public Waters. Nebraska Game and Parks Commission, Lincoln, Nebraska. 71 pp.

Nebraska Game and Parks Commission. 1996. Platte River Fish Kill Reports from 1974 through 1996. NGPC, Fisheries Division, Lincoln, Nebraska. Unpublished tabulation.

Pain, Deborah J. 1996. Lead in Waterfowl. In: Environmental Contaminants in Wildlife - Interpreting Tissue Concentrations. W. Nelson Beyer, Gary H. Heinz, and Amy W Redmon-Norwood (Eds.), SETAC Special Publication Series, Lewis Publishers, Boca Raton, Florida. Pp. 251-264.

Pais, István, and J. Benton Jones. 1997. The Handbook of Trace Elements. St. Lucie Press, Boca Raton, Florida. 223 pp.

Parker, Raymond L. 1967. Data of Geochemistry Sixth Edition, Chapter D. Composition of the Earth's Crust. Geological Survey Professional Paper 440D. United States Government Printing Office, Washington, D.C. 19 pp.

Peckenpaugh, J,M., and J.T. Dugan. 1983. Hydrogeology of the Central Platte and Lower Loup Natural Resource Districts, Nebraska. Water-Resources Investigations Report 83-4219. U.S. Geological Survey, Lincoln, Nebraska. 100 pp.

Peckinpaugh, J.M., J.T. Dugan, R.A. Kern, and W.J. Schroeder. 1987. Hydrogeology of the Tri–Basin and Parts of the Lower Republican and Central Platte Natural Resources Districts, Nebraska. U.S. Geological Survey Water Resources Investigations Report 87–4176. USGS, Lincoln, Nebraska. 117 pp.

Peterson, David A., William E, Jones, and Anthony G. Morton. 1988. Reconnaissance Investigation of Water Quality, Bottom Sediment, and Biota associated with Irrigation Drainage in the Kendrick Reclamation Project Area, Wyoming, 1986-87. U.S. Geological Survey Water-Resources Investigations Report 87-4255, USGS, Cheyenne, Wyoming. 57 pp.

Porcella, Donald B. 1994. Mercury in the Environment: Biogeochemistry. In: Mercury Pollution: Integration and Synthesis, Carl J. Watras and John W. Huckabee (eds.). Lewis Publishers, Boca Raton, Florida. 727 pp.

Public Health Service. 1951. North Platte River Basin Water Pollution Investigation. PHS, Kansas City, Missouri. 149 pp.

Reckhow, Kenneth H., and Steven C. Chapra. 1983. Engineering Approaches to Lake Management. Volume 1: Data Analysis and Empirical Modeling. Ann Arbor Science, Butterworth Publishers, Woburn, Massachusetts. 340 pp.

Rosenfield, Irene, and Orville A. Beath. 1964. Selenium – Geobotany, Biochemistry, Toxicity, and Nutrition. Academic Press, New York, N.Y. 411 pp.

Sartoris, J.J., J.F. LaBounty, S.G. Campbell, and J.R. Boehmke. 1981. Limnology of the Upper North Platte Reservoir System, Wyoming. Report REC-ERC-81-10, Bureau of Reclamation, Denver, Colorado. 129 pp.

Schmitt, Christopher J., Jim L. Zajicek, and Michael A. Ribick. 1985. National Pesticide Monitoring Program: Residues of Organochlorine Chemicals in Freshwater Fish, 1980-81. Arch. Environ. Contam. Toxicol. 14:225-260.

Schmitt, Christopher J., and William G. Brumbaugh. 1990. National Contaminant Biomonitoring Program: Concentrations of Arsenic, Cadmium, Copper, Lead, Mercury, Selenium, and Zinc in U.S. Freshwater Fish, 1976-84. Arch. Environ. Contam. Toxicol. 19:731-747.

Seavy, Louis M., and Frank L. Illk. 1953. Sedimentation Surveys of Pathfinder and Seminoe Reservoirs, North Platte River, Wyoming. Sedimentation Section Report, Bureau of Reclamation, Denver, Colorado. 58 pp.

See, Randolph B., David L. Naftz, David A. Peterson, James G. Crock, James A. Erdman, R.C. Severson, Pedro Ramirez Jr., and Joni A. Armstrong. 1992. Detailed Study of Selenium in Soil, Representative Plants, Water, Bottom Sediment, and Biota in the Kendrick Reclamation Project, Wyoming, 1988-90. U.S. Geological Survey Water-Resources Investigations Report 91-4131, USGS, Cheyenne, Wyoming. 142 pp.

Severson, R.C., K.C. Stewart, and Thelma F. Harms. 1991. Partitioning of Elements between Two Sediment Fractions in Samples from Nineteen Areas of the Western United States. U.S. Geological Survey Open-File Report 91-381, USGS Denver, Colorado. 18 pp.

Shacklette, Hansford T., and Josephine G. Boerngen. 1984. Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States. U.S. Geological Survey Professional Paper 1270. United States Government Printing Office, Washington, D.C. 105 pp.
Sorensen, Elsa M. 1991. Metal Poisoning in Fish. CRC Press, Boca Raton, Florida. 374 pp.

Sheppard, John R. 1959. Observation of Sedimentation in Pathfinder Reservoir, Wyoming. Sedimentation Section Report, Bureau of Reclamation, Denver, Colorado. 11 pp. + 2 tables + 15 Figures

Sinokrot, Bashar A., John S. Gulliver, and Ruochuan Gu. 1997. Impacts of In-Stream Flow Requirements on Water Temperature in the Central Platte River. In: Waterpower '97: proceedings of the International Conference on Hydropower. Daniel J. Mahoney (ed.), American Society of Civil Engineers, New York, New York. Vol 1, pages 1-9.

Sorensen, Elsa M. 1991. Metal Poisoning in Fish. CRC Press, Boca Raton, Florida. 374 pp.

SPSS, Inc. 1999. SYSTAT 9 Users Manual, SPSS, Inc., Chicago, Illinois. 6 Vols.

Stansbury, John, David Admiraal, Eric Dove, Kyle Hoagland, and John Holz. 2002a. Water Quality Modeling of Lake Ogallala Using the WASP-5 Model. A Study Conducted for the Nebraska Game and Parks Commission. University of Nebraska, Lincoln, Nebraska. 25 pp. + attached tables & figures + App.

Stansbury, John, David Admiraal, Eric Dove, Kyle Hoagland, and John Holz. 2002b. Water Circulation Modeling of Lake Ogallala Using the RMA-2 Model. A Study Conducted for the Nebraska Game and Parks Commission. University of Nebraska, Lincoln, Nebraska. 10 pp. +13 Figs. 78 pp. + App.

Steele, G.V., J.C. Cannia, S.S. Sibray, and V.L. McGuire. 2005. Age and Quality of Ground Water and Sources of Nitrogen in the Surficial Aquifers in Pumpkin Creek Valley, Western Nebraska, 2000. U.S. Geological Survey Scientific Investigations Report 2005-5157, USGS, Lincoln, Nebraska. 68 pp.

Thompson, David R. 1996. Mercury in Birds and Terrestrial Mammals. In: Environmental Contaminants in Wildlife - Interpreting Tissue Concentrations. W. Nelson Beyer, Gary H. Heinz, and Amy W Redmon-Norwood (Eds.), SETAC Special Publication Series, Lewis Publishers, Boca Raton, Florida. Pp. 341-356.

U.S. Environmental Protection Agency. 1976. Report on C.W. McConaughy Reservoir, Keith County, Nebraska. EPA Region VII. Working Paper 559. National Eutrophication Survey, Corvallis Environmental Research Laboratory, Corvallis, Oregon, and Environmental Monitoring & Support Laboratory, Las Vegas, Nevada.
12 pp. + 5 App.

U.S. Environmental Protection Agency. 1977. Report on Seminoe Reservoir, Carbon County, Wyoming. Working Paper #890, EPA, National Eutrophication Survey, Corvallis, Oregon.
46 pp.

U.S. Environmental Protection Agency. 1985. Ambient Water Quality Criteria for Ammonia – 1984. EPA 440/5-85-001. EPA, Office of Water, Regulations and Standards Criteria and Standards Division, Washington, DC 217 pp.

U.S. Environmental Protection Agency. 1986. Quality Criteria for Water 1986. EPA 440/5-86-001. EPA, Office of Water, Regulations and Standards, Washington, D.C. Unnumbered.

U.S. Environmental Protection Agency. 1989. U.S. EPA IRIS Substance file - Uranium, soluble salts. EPA website: http://www.epa.gov/ngispgm3/iris/subst/0421.htm

U.S. Environmental Protection Agency. 1999. 1999 Update of Ambient Water Quality Criteria for Ammonia – Supersedes 1998 Update. EPA-822-R-99-014. EPA, Office of Water, Washington, DC 147 pp.

U.S. Environmental Protection Agency. 2001. Ambient Water Quality, Criteria Recommendations Information Supporting the Development of State and Tribal Nutrient Criteria, Lakes and Reservoirs in Nutrient Ecoregion V. EPA 822-B-01-010, EPA, Office of Water, Washington, DC. 29 pp. + 3 App.

U.S. Environmental Protection Agency. 2002. National Recommended Water Quality Criteria: 2002. EPA-822-R-02-047, EPA, Office of Water and Office of Science and Technology, Washington, DC. 30 pp. + 3 App.

U.S. Environmental Protection Agency. 2003. National Primary Drinking Water Standards. EPA, Office of Water, Washington, D.C. 6 pp. (http://www.epa.gov/safewater/consumer/pdf/mcl.pdf)

U.S. Environmental Protection Agency. 2004. The Incidence and Severity of Sediment Contamination in Surface Waters of the United States. EPA-823-R-04-007, EPA, Office of Science and Technology, Washington, D.C. Numbered by section.

U.S. Fish and Wildlife Service. 1997. Biological Opinion on the Federal Energy Commission's Preferred Alternative for the Kingsley Dam Project (Project No. 1417) and North Platte/Keystone Dam Project (Project No. 1835), USFWS, Grand Island, Nebraska.

U.S. Geological Survey. 1981. Quality of Water Branch Memorandum No. 81.12 - Subject: WATER QUALITY–Trace Metals; Questionable Values of Dissolved and Total Selenium. USGS, Reston, Virginia.

U.S. Geological Survey. 2001. Water Resources Applications Software – DOTABLES: Dissolved oxygen saturation tables. *http://water.usgs.gov/software/dotables.html* Accessed on 12/8/2005.

Van Velson, Rodney C. 1978. The Lake McConaughy Rainbow...Life History and a Management Plan for the North Platte River Valley. Nebraska Technical Series No. 2, Nebraska Game and Parks Commission, Lincoln, Nebraska. 83 pp.

Verstraeten, I.M., G.V. Steele, J.C. Cannia, D.E. Hitch, K.G. Scripter, J.K. Böhlke, T.F. Kraemer, and J.S. Stanton. 2001. Interaction of Surface Water and Ground Water in the Dutch Flats Area, Western Nebraska, 1995-99. Water-Resources Investigations Report 01-4070, USGS, Lincoln, Nebraska. 56 pp.

Verstraeten, I.M., S.S. Sibray, J.C. Cannia, and D.Q. Tanner. 1995. Reconnaissance of Ground-Water Quality in the North Platte Natural Resources District, Western Nebraska, June-July 1991. U.S. Geological Survey Water-Resources Investigations Report 94-4057, USGS, Lincoln, Nebraska. 114 pp.

Vollenweider, R.A., and P.J. Dillon. 1974. The application of the phosphorus loading concept to eutrophication research. National Research Council of Canada publication number 13690, Canada Centre for Inland Waters, Burlington, Ontario (cited in EPA, 1976).

Wahl, Frank E. 1981. Soil Survey of Gosper County, Nebraska. U.S. Department of Agriculture, Soil Conservation Service (now, Natural Resources Conservation Service) and University of Nebraska, Conservation and Survey Division, Lincoln, Nebraska. 123 pp. + 35 maps + Index.

Wahl, Frank E., Jay Wilson, and Steve Scheinost. 1984. Soil Survey of Kearney County, Nebraska. U.S. Department of Agriculture, Soil Conservation Service (now, Natural Resources Conservation Service) and University of Nebraska, Conservation and Survey Division, Lincoln, Nebraska. 157 pp. + 41 maps + Index.

Water Quality Control Division. 2004. Status of Water Quality in Colorado – 2004: The Update to the 2002 305(b) Report. Colorado Department of Health and Environment, WQCD, Denver, Colorado. 25 pp. + 4 App.

Waters, Thomas. 1995. Sediment in Streams: Sources, Biological Effects and Control. Monograph 7, American Fisheries Society, Bethesda, Maryland. 251 pp.

WES. 1995. CE-QUAL-R1: A numerical one-dimensional model of reservoir water quality; User's Manual, Instruction Report E-82-1 (Revised Edition), Environmental Laboratory, U.S. Army Engineers Waterways Experiment Station, Vicksburg, Mississippi.

Wilson, Erika C., and Wayne C. Hubert, and Stanley H. Anderson. 1993. Nesting and Foraging of Least Terns on Sand Pits in Central Nebraska. The Southwestern Naturalist 38(1):9-14.

Wright, Winfield G. 1999. Oxidation and Mobilization of Selenium by Nitrate in Irrigation Drainage. J. Environ. Quality. 28:1182-1187.

Wyoming Department of Environmental Quality. 2004. Wyoming's 2004 305(b) State Water Quality Assessment Report and 2004 303(d) List of Waters Requiring TMDLs. DEQ, Cheyenne, Wyoming. 91 pp

Wyoming Department of Environmental Quality. 2002. Wyoming's 2002 305(b) State Water Quality Assessment Report and 2002 303(d) List of Waters Requiring TMDLs. DEQ, Cheyenne, Wyoming. 84 pp.

Wyoming Department of Environmental Quality. 2000. Wyoming's 2000 305(b) State Water Quality Assessment Report. DEQ, Cheyenne, Wyoming. 7 pp. + 4 App.

Wyoming Game and Fish Department. 2004. Letter of September 13, 2004, to Platte River EIS Office. WGFD, Casper, Wyoming. 18 pp.

Yahnke, James. 1990. A Mathematical Simulation of the Temperature and Dissolved Oxygen Regime of Lake McConaughy, Nebraska. Special Report prepared in cooperation with the Nebraska Game and Parks Commission, Central Nebraska Public Power and Irrigation District, Nebraska Public Power District, and U.S. Fish and Wildlife Service. Bureau of Reclamation, Grand Island, Nebraska. 27 pp. + 5 App.

Yahnke, Jim. 2001. Impacts of Alternatives and Program Elements on Water Quality of the Platte River. Technical Report of the Platte River EIS Team, U.S. Department of the Interior, Platte River EIS Office, Denver, Colorado. 51 pp. + Attach.

Zander, Bruce. 1996. Review of Instream Flows and Ambient Water Quality Temperature Standards of the Platte River Downstream of the Kingsley Dam Project (FERC 1417) and Keystone Diversion Dam Project (FERC 1835), Second Updated Report. U.S. Environmental Protection Agency, Region VIII, Denver, Colorado.

ATTACHMENT A

The National Criteria For Ammonia in Fresh Water

Ammonia Toxicity Criteria

EPA (1985) developed ammonia criteria based solely on concentrations of unionized ammonia. Formulae were provided to calculate the unionized ammonia fraction from the total ammonia (unionized ammonia + ammonium) concentrations. The unionized ammonia-ammonium equilibrium is temperature and pH dependent. The ammonia criterion was to be calculated from formulae based on temperature and pH as well.

EPA (1998) updated EPA (1985) and developed acute and chronic ammonia criteria based on total ammonia concentrations. The change in philosophy was based on evidence that the ammonium ion also contributed to the toxicity of ammonia. However, it was also noted that unionized ammonia was about 100 times as toxic as the ammonium ion alone. The problem is that neither occurs alone and it was difficult to isolate the effects of the individual forms of ammonia. The total ammonium criteria were strictly pH dependent.

EPA (1999) superseded EPA (1998) and further updated EPA (1984). A major concern in EPA (1999) was the effect of temperature on the chronic toxicity of ammonia. There was also concern over the averaging period for the chronic ammonia criterion. The final 1999 criteria are shown below.

As was done in the 1998 update, there is no temperature correction for the acute ammonia criterion, but there are separate criteria depending on the presence or absence of salmonids, which are more sensitive to ammonia than other families of fishes. This is a holdover from the 1984 criteria.

There are also 2 chronic ammonia criteria. The first, and more restrictive criterion, is applicable when early life stages (eggs and fry) of fish are present. The second is applicable when the early life stages of fish are absent.

There is considerable evidence that ammonia is less toxic at lower temperatures. For this reason, there are lower limits on the temperature adjustments for the chronic ammonia criteria.

In 2004, EPA announced another review of the ammonia criteria (EPA, 2004). The review was prompted by more recent studies that indicated that the 1999 criteria may not protect certain species of freshwater mussels (Augsberger, *et al*., 2003). This review is currently ongoing.

The National Criterion For Ammonia in Fresh Water

The available data for ammonia, evaluated using the procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses", indicate that, except possibly where an unusually sensitive species is important at a site, freshwater aquatic life should be protected if both of the following conditions are satisfied for the temperature, T, and pH of the waterbody:

1. The one-hour average concentration of total ammonia nitrogen (in mg N/L) does not exceed, more than once every three years on the average, the CMC (acute criterion) calculated using the following equations. Where salmonid fish are present:

$$CMC = \frac{0.275}{1 + 10^{7.204-pH}} + \frac{39.0}{1 + 10^{pH-7.204}}$$

Or where salmonid fish are not present:

$$CMC = \frac{0.411}{1 + 10^{7.204-pH}} + \frac{58.4}{1 + 10^{pH-7.204}}$$

2A. The thirty-day average concentration of total ammonia nitrogen (in mg N/L) does not exceed, more than once every three years on the average, the CCC (chronic criterion) calculated using the following equations.
When fish early life stages are present:

$$CCC = \left(\frac{0.0577}{1 + 10^{7.688-pH}} + \frac{2.487}{1 + 10^{pH-7.688}} \right) \cdot MIN(2.85,\ 1.45 \cdot 10^{0.028 \cdot (25-T)})$$

When fish early life stages are absent:

$$CCC = \left(\frac{0.0577}{1 + 10^{7.688-pH}} + \frac{2.487}{1 + 10^{pH-7.688}} \right) \cdot 1.45 \cdot 10^{0.028 \cdot (25-MAX(T,7))}$$

2B. In addition, the highest four-day average within the 30-day period should not exceed 2.5 times the CCC.

Tables showing the calculated CMC and CCC values for a range of pH and temperature (CCC only) appear below.

pH-Dependent Values of the CMC (Acute Criterion)		
CMC, mg N/L		
pH	Salmonids Present	Salmonids Absent
6.5	32.60	48.80
6.6	31.30	46.80
6.7	29.80	44.60
6.8	28.10	42.00
6.9	26.20	39.10
7.0	24.10	36.10
7.1	22.00	32.80
7.2	19.70	29.50
7.3	17.50	26.20
7.4	15.40	23.00
7.5	13.30	19.90
7.6	11.40	17.00
7.7	9.65	14.40
7.8	8.11	12.10
7.9	6.77	10.10
8.0	5.62	8.40
8.1	4.64	6.95
8.2	3.83	5.72
8.3	3.15	4.71
8.4	2.59	3.88
8.5	2.14	3.20
8.6	1.77	2.65
8.7	1.47	2.20
8.8	1.23	1.84
8.9	1.04	1.56
9.0	0.885	1.32

Temperature and pH-Dependent Values of the CCC (Chronic Criterion) for Fish Early Life Stages Present										
CCC for Fish Early Life Stages Present, mg N/L										
pH	Temperature (°C)									
	0	14	16	18	20	22	24	26	28	30
6.5	6.67	6.67	6.06	5.33	4.68	4.12	3.62	3.18	2.8	2.46
6.6	6.57	6.57	5.97	5.25	4.61	4.05	3.56	3.13	2.75	2.42
6.7	6.44	6.44	5.86	5.15	4.52	3.98	3.5	3.07	2.7	2.37
6.8	6.29	6.29	5.72	5.03	4.42	3.89	3.42	3	2.64	2.32
6.9	6.12	6.12	5.56	4.89	4.3	3.78	3.32	2.92	2.57	2.25
7.0	5.91	5.91	5.37	4.72	4.15	3.65	3.21	2.82	2.48	2.18
7.1	5.67	5.67	5.15	4.53	3.98	3.5	3.08	2.7	2.38	2.09
7.2	5.39	5.39	4.9	4.31	3.78	3.33	2.92	2.57	2.26	1.99
7.3	5.08	5.08	4.61	4.06	3.57	3.13	2.76	2.42	2.13	1.87
7.4	4.73	4.73	4.3	3.78	3.32	2.92	2.57	2.26	1.98	1.74
7.5	4.36	4.36	3.97	3.49	3.06	2.69	2.37	2.08	1.83	1.61
7.6	3.98	3.98	3.61	3.18	2.79	2.45	2.16	1.9	1.67	1.47
7.7	3.58	3.58	3.25	2.86	2.51	2.21	1.94	1.71	1.5	1.32
7.8	3.18	3.18	2.89	2.54	2.23	1.96	1.73	1.52	1.33	1.17
7.9	2.8	2.8	2.54	2.24	1.96	1.73	1.52	1.33	1.17	1.03
8.0	2.43	2.43	2.21	1.94	1.71	1.5	1.32	1.16	1.02	0.897
8.1	2.1	2.1	1.91	1.68	1.47	1.29	1.14	1	0.879	0.773
8.2	1.79	1.79	1.63	1.43	1.26	1.11	0.973	0.855	0.752	0.661
8.3	1.52	1.52	1.39	1.22	1.07	0.941	0.827	0.727	0.639	0.562
8.4	1.29	1.29	1.17	1.03	0.906	0.796	0.7	0.615	0.541	0.475
8.5	1.09	1.09	0.99	0.87	0.765	0.672	0.591	0.52	0.457	0.401
8.6	0.92	0.92	0.836	0.735	0.646	0.568	0.499	0.439	0.386	0.339
8.7	0.778	0.778	0.707	0.622	0.547	0.48	0.422	0.371	0.326	0.287
8.8	0.661	0.661	0.601	0.528	0.464	0.408	0.359	0.315	0.277	0.244
8.9	0.565	0.565	0.513	0.451	0.397	0.349	0.306	0.269	0.237	0.208
9.0	0.486	0.486	0.442	0.389	0.342	0.300	0.264	0.232	0.204	0.179

Temperature and pH-Dependent Values of the CCC (Chronic Criterion) for Fish Early Life Stages Absent										
CCC for Fish Early Life Stages (ELS) Absent, mg N/L										
pH	Temperature (°C)									
	0-7	8	9	10	11	12	13	14	15*	16*
6.5	10.80	10.10	9.51	8.92	8.36	7.84	7.35	6.89	6.46	6.06
6.6	10.70	9.99	9.37	8.79	8.24	7.72	7.24	6.79	6.36	5.97
6.7	10.50	9.81	9.20	8.62	8.08	7.58	7.11	6.66	6.25	5.86
6.8	10.20	9.58	8.98	8.42	7.90	7.40	6.94	6.51	6.10	5.72
6.9	9.93	9.31	8.73	8.19	7.68	7.20	6.75	6.33	5.93	5.56
7.0	9.60	9.00	8.43	7.91	7.41	6.95	6.52	6.11	5.73	5.37
7.1	9.20	8.63	8.09	7.58	7.11	6.67	6.25	5.86	5.49	5.15
7.2	8.75	8.20	7.69	7.21	6.76	6.34	5.94	5.57	5.22	4.90
7.3	8.24	7.73	7.25	6.79	6.37	5.97	5.60	5.25	4.92	4.61
7.4	7.69	7.21	6.76	6.33	5.94	5.57	5.22	4.89	4.59	4.30
7.5	7.09	6.64	6.23	5.84	5.48	5.13	4.81	4.51	4.23	3.97
7.6	6.46	6.05	5.67	5.32	4.99	4.68	4.38	4.11	3.85	3.61
7.7	5.81	5.45	5.11	4.79	4.49	4.21	3.95	3.70	3.47	3.25
7.8	5.17	4.84	4.54	4.26	3.99	3.74	3.51	3.29	3.09	2.89
7.9	4.54	4.26	3.99	3.74	3.51	3.29	3.09	2.89	2.71	2.54
8.0	3.95	3.70	3.47	3.26	3.05	2.86	2.68	2.52	2.36	2.21
8.1	3.41	3.19	2.99	2.81	2.63	2.47	2.31	2.17	2.03	1.91
8.2	2.91	2.73	2.56	2.40	2.25	2.11	1.98	1.85	1.74	1.63
8.3	2.47	2.32	2.18	2.04	1.91	1.79	1.68	1.58	1.48	1.39
8.4	2.09	1.96	1.84	1.73	1.62	1.52	1.42	1.33	1.25	1.17
8.5	1.77	1.66	1.55	1.46	1.37	1.28	1.20	1.13	1.06	0.99
8.6	1.49	1.40	1.31	1.23	1.15	1.08	1.01	0.951	0.892	0.836
8.7	1.26	1.18	1.11	1.04	0.976	0.915	0.858	0.805	0.754	0.707
8.8	1.07	1.01	0.944	0.885	0.829	0.778	0.729	0.684	0.641	0.601
8.9	0.917	0.860	0.806	0.756	0.709	0.664	0.623	0.584	0.548	0.513
9.0	0.790	0.740	0.694	0.651	0.610	0.572	0.536	0.503	0.471	0.442
* At 15°C and above, the criterion for fish ELS absent is the same as the criterion for fish ELS present.										

References

Augsberger, Tom, Anne E. Keller, Marsha C. Black, W. Gregory Cope, and F. James Dwyer. 2003. Water Quality Guidance For Protection Of Freshwater Mussels (Unionidae) From Ammonia Exposure. Environmental Toxicology and Chemistry 22(11):2569–2575.

U.S. Environmental Protection Agency. 1985. Ambient Water Quality Criteria for Ammonia – 1984. EPA 440/5-85-001. EPA, Office of Water, Regulations and Standards Criteria and Standards Division, Washington, DC 217 pp.

U.S. Environmental Protection Agency. 1998. 1998 Update of Ambient Water Quality Criteria for Ammonia. EPA 822-R-98-008. EPA, Office of Water, Washington, DC 148 pp.

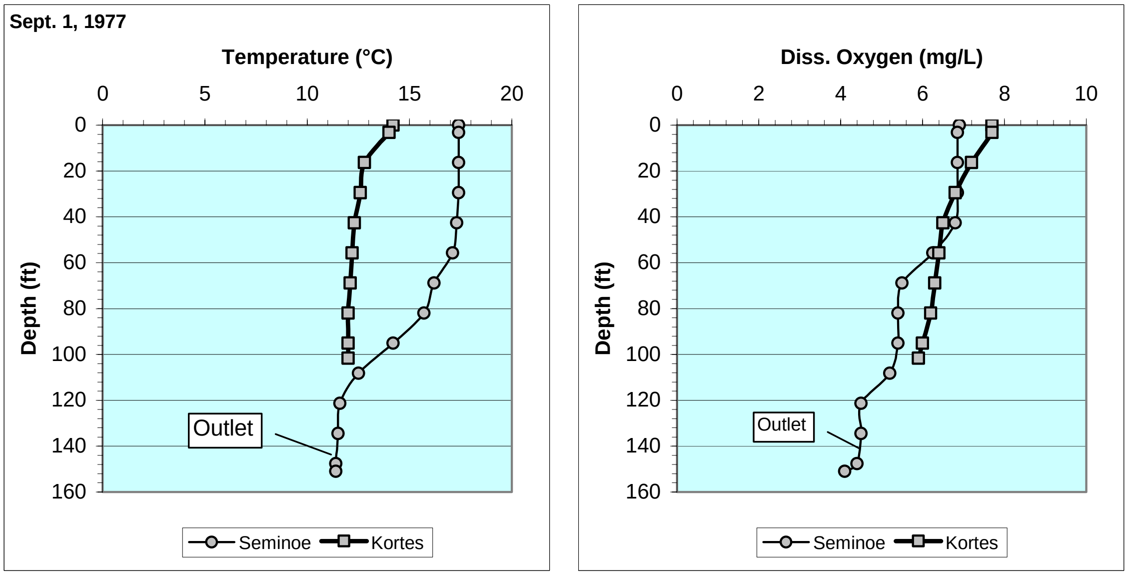
U.S. Environmental Protection Agency. 1999. 1999 Update of Ambient Water Quality Criteria for Ammonia – Supersedes 1998 Update. EPA-822-R-99-014. EPA, Office of Water, Washington, DC 147 pp.

U.S. Environmental Protection Agency. 2004. Notice of Intent To Re-Evaluate the Aquatic Life Ambient Water Quality Criteria for Ammonia. Fact Sheet; July 2004. Downloaded on November 18, 2005 from:
http://www.epa.gov/waterscience/criteria/ammonia/re-eval-fs.htm

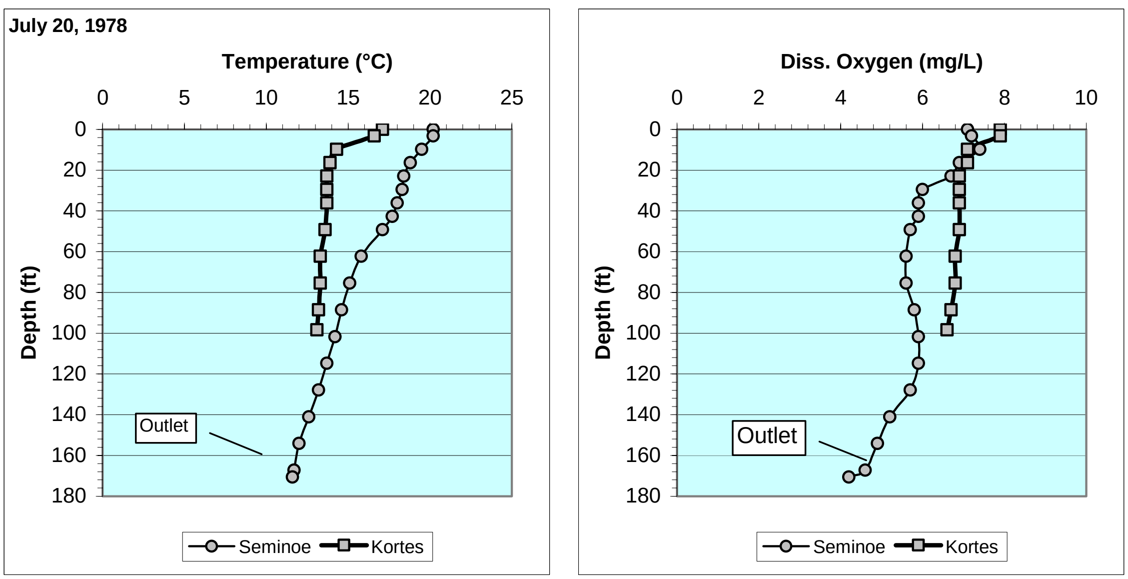
ATTACHMENT B

SEMINOE AND KORTES RESERVOIR

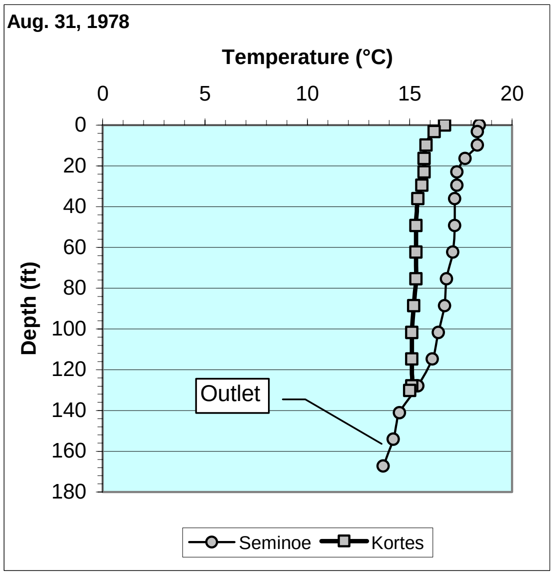
TEMPERATURE AND DISSOLVED OXYGEN PROFILES

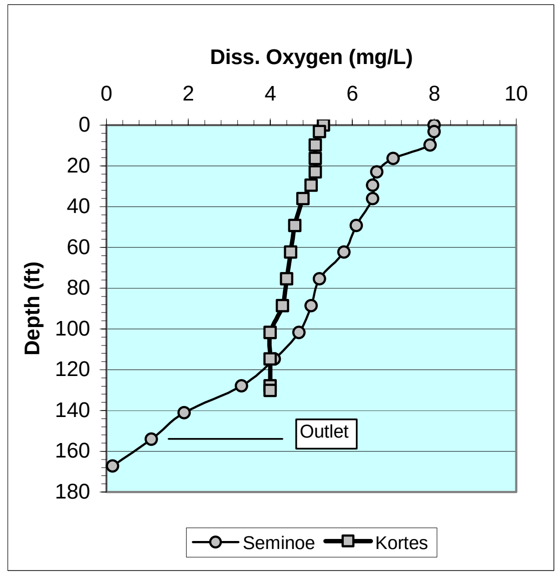


Most of Kortes Reservoir temperature profile, except at surface, is the same as Seminoe Reservoir outlet temperature, but DO is higher, indicating reaeration

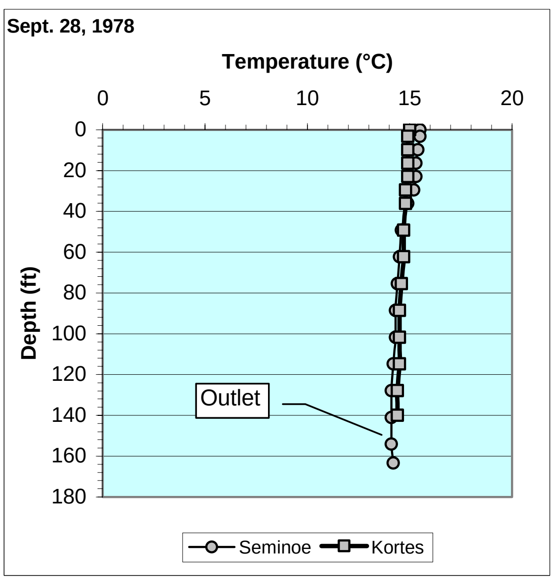


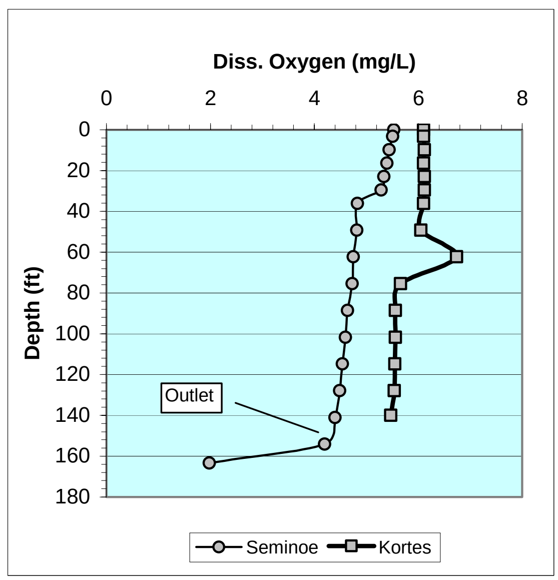
Most of Kortes Reservoir temperature profile, except at surface, is the same as Seminoe Reservoir outlet temperature, but DO is higher, indicating reaeration



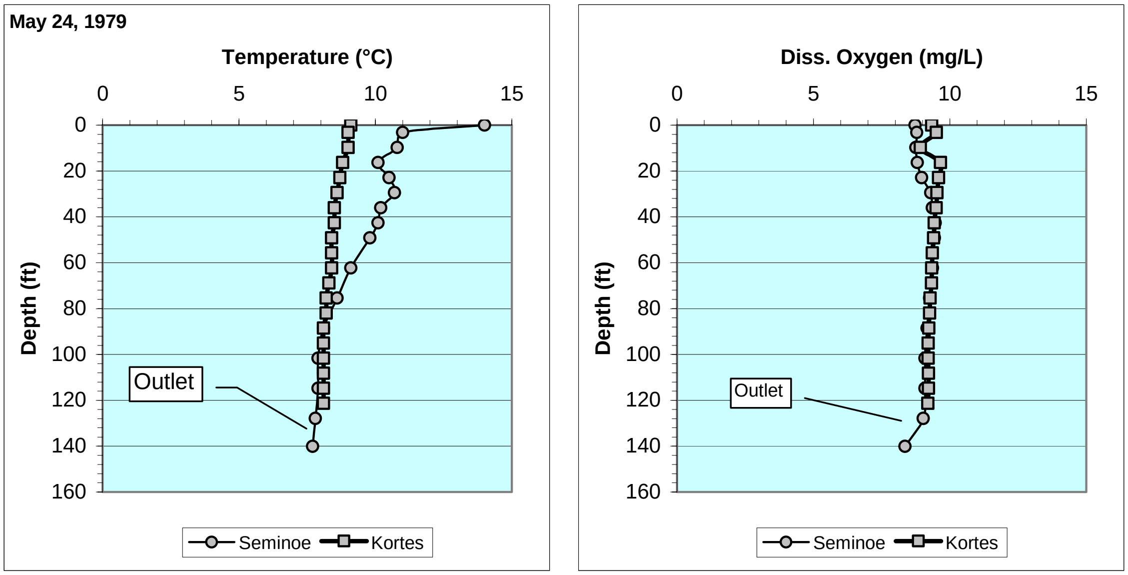


Most of Kortes Reservoir temperature profile, except at surface, is the same as Seminoe Reservoir outlet temperature, but DO is higher, indicating some reaeration occurred, but DO is still less than in the upper layers of the Seminoe Reservoir profile

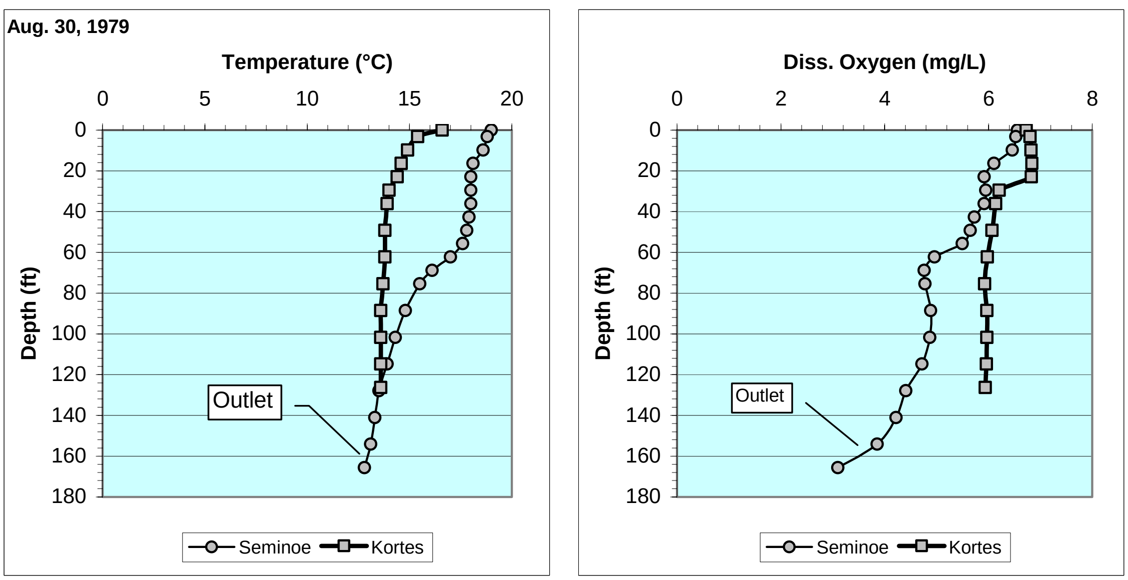




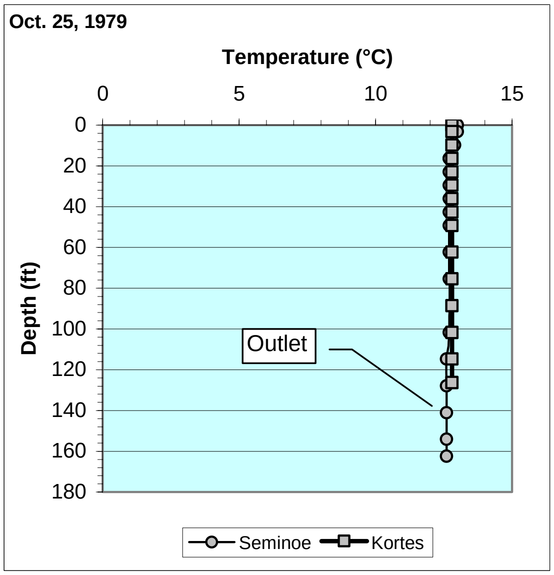
Most of Kortes Reservoir temperature profile, except at surface, is the same as Seminoe Reservoir outlet temperature, but DO is higher, indicating some reaeration occurred; entire Kortes profile is higher than any layers of the Seminoe Reservoir DO profile. The DO peak at 60 feet in Kortes appears to indicate a photosynthetic maximum.

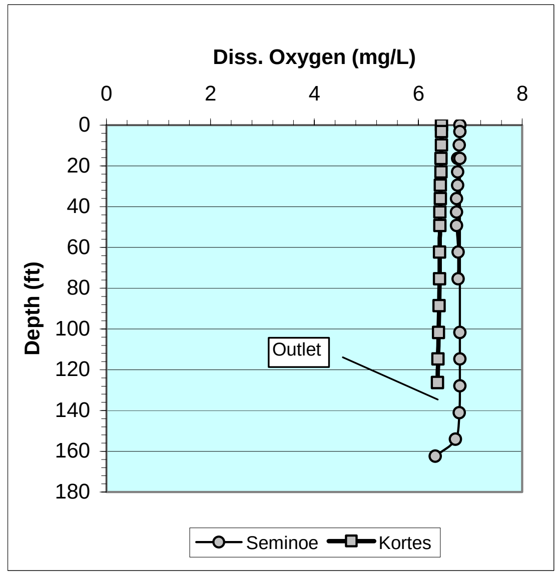


Most of Kortes Reservoir temperature profile, except at surface, is the same as Seminoe Reservoir outlet temperature; DO profiles are essentially identical

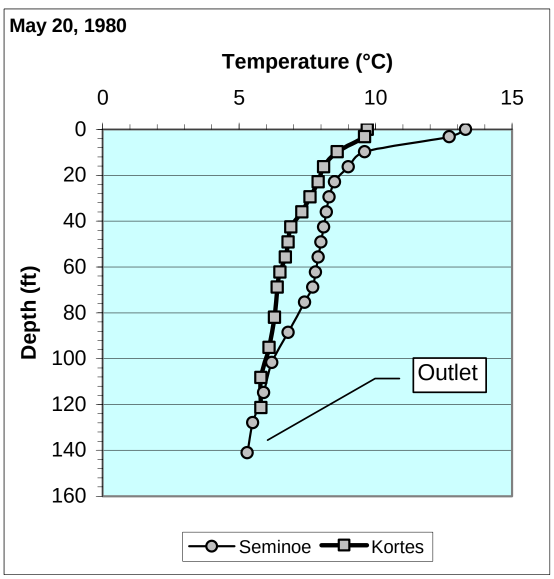


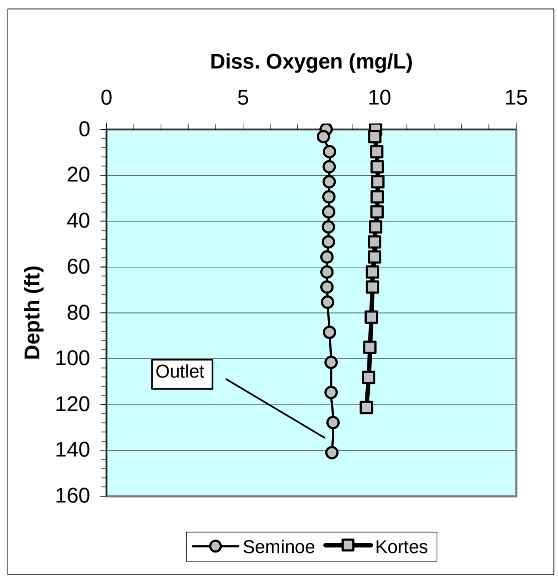
Most of Kortes Reservoir temperature profile, except at surface, is the same as Seminoe Reservoir outlet temperature, but DO is higher, indicating reaeration



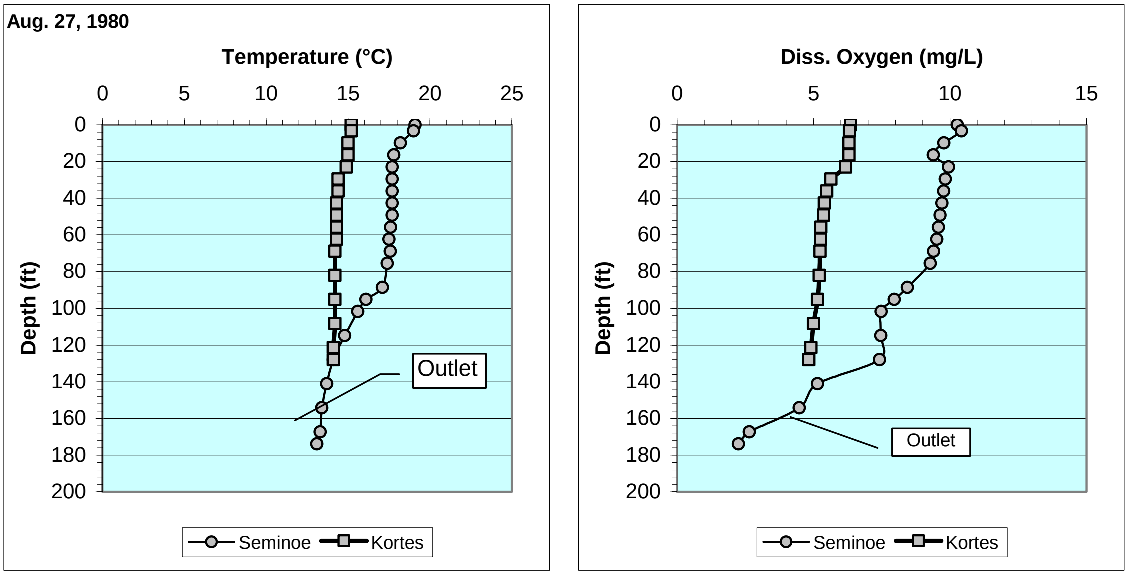


Most of Kortes Reservoir temperature profile, except at surface, is the same as Seminoe Reservoir outlet temperature, but DO is lower than all but the bottom of the Seminoe Reservoir profile (Kortes profile measured on previous day)





Kortes Reservoir somewhat cooler than Seminoe through most of the profile; DO somewhat higher in Kortes than Seminoe, indicating aeration

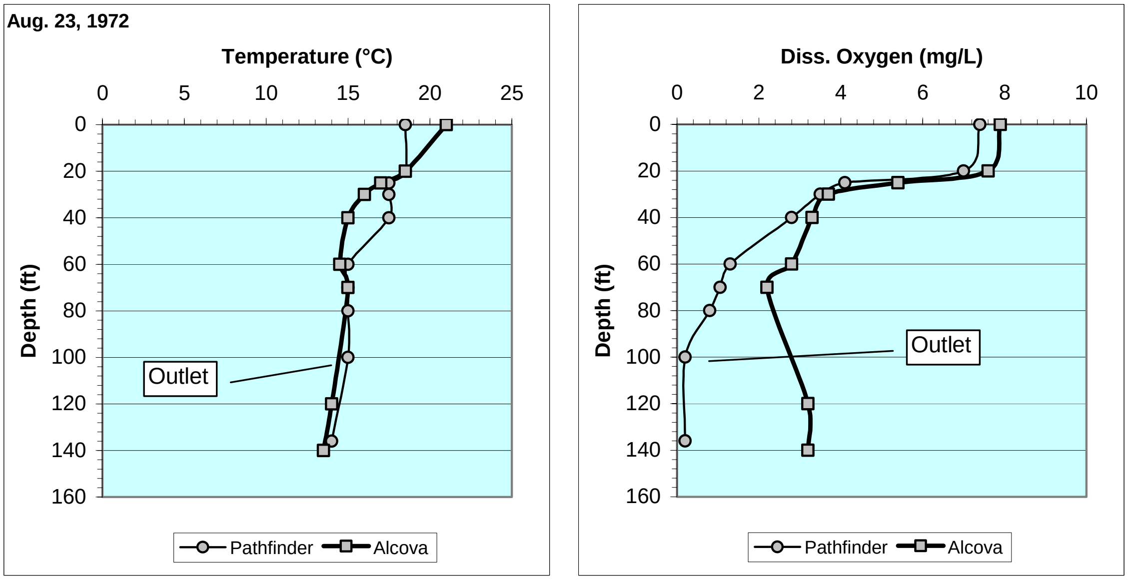


Most of Kortes Reservoir temperature and DO profiles are similar to those of Seminoe Reservoir outlet, indicating little change in Kortes Reservoir or in the intervening river reach

ATTACHMENT C

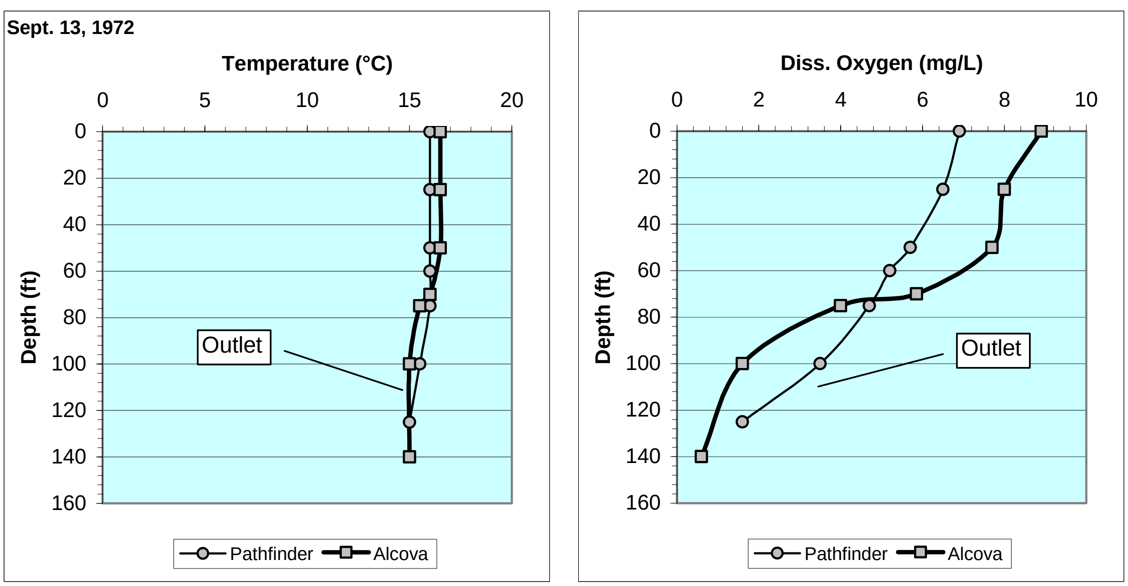
PATHFINDER AND ALCOVA RESERVOIR

TEMPERATURE AND DISSOLVED OXYGEN PROFILES

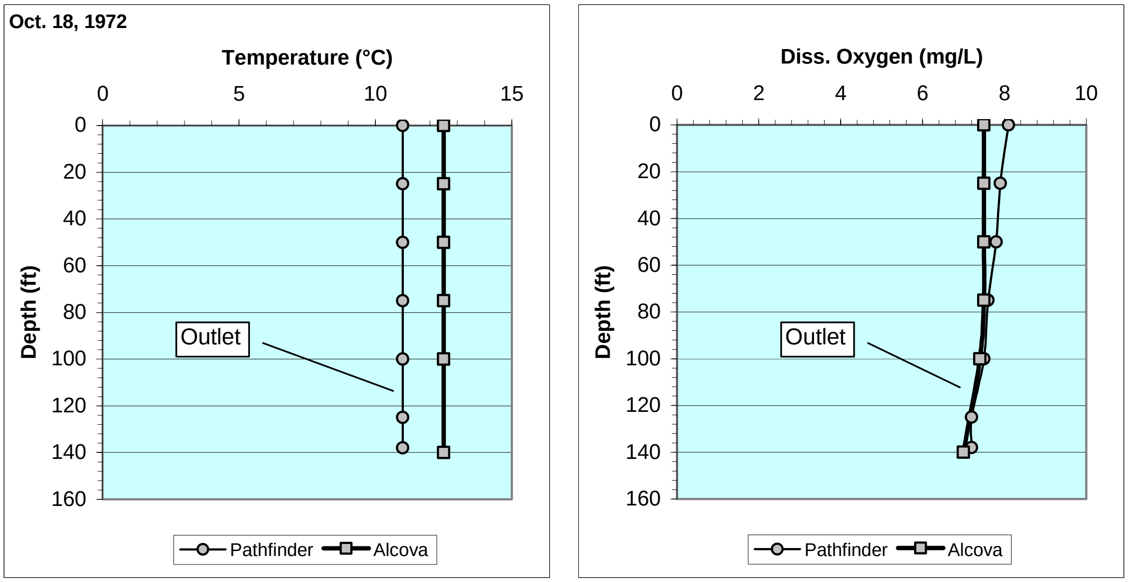


Pathfinder outlet elevation is shown on each plot.
Alcova outlet always at either 10 or 20 foot depth.

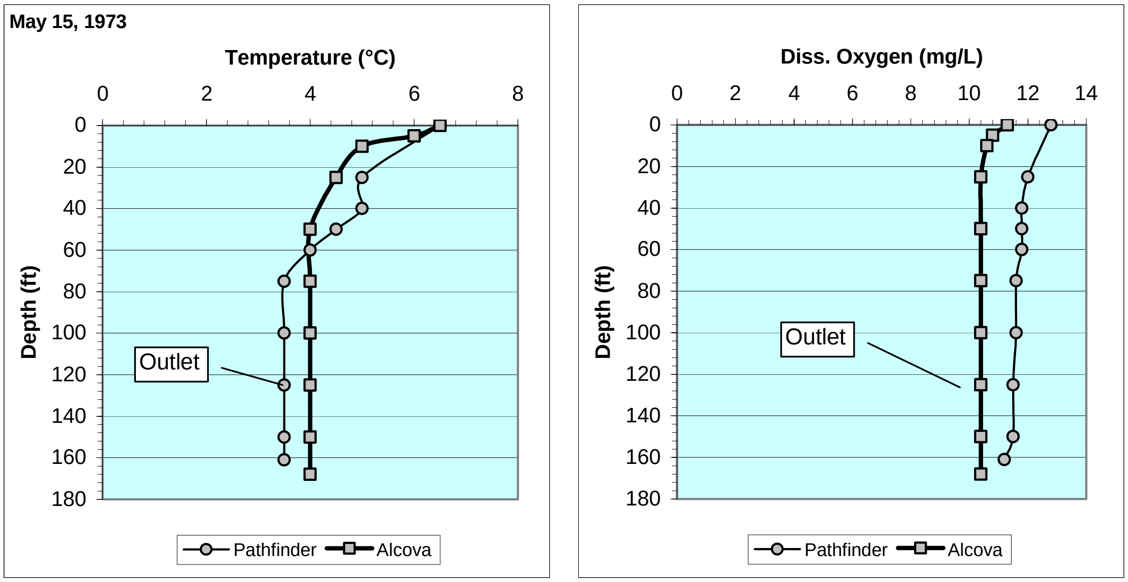
Temperature profiles are similar in the 2 reservoirs.
Alcova hypolimnion better aerated than Pathfinder.



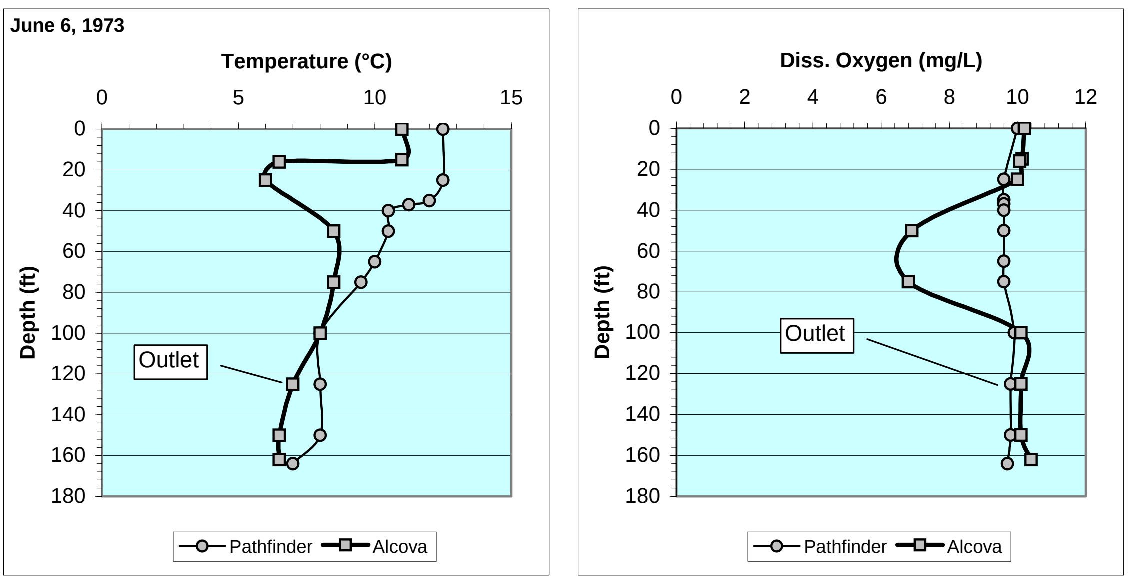
Alcova and Pathfinder temperatures profiles are again similar, but slight stratification in Alcova. Slight stratification has marked effect on DO distribution in Alcova. Alcova well-aerated near the surface, poorly aerated at depth. Distinct break in DO distribution coincides with elevation of slight temperature change. Pathfinder Reservoir appears to be at the onset of the process of mixing and reaeration.



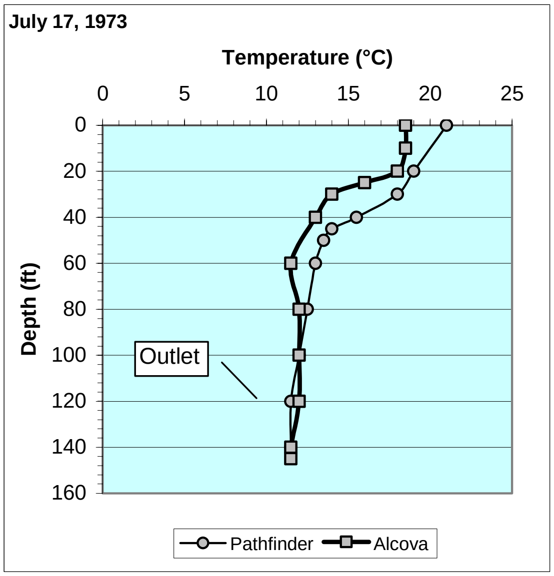
Both reservoirs are isothermal and well mixed and well aerated.

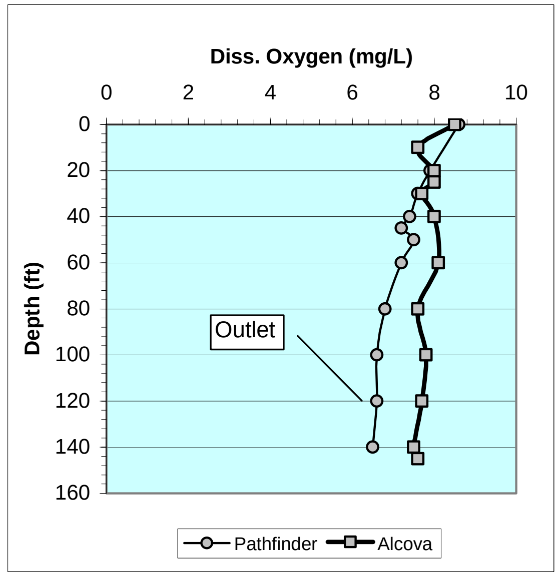


Slightly stronger thermal stratification in Pathfinder, but weak in both.
Both reservoirs well aerated, but Pathfinder slightly more so.

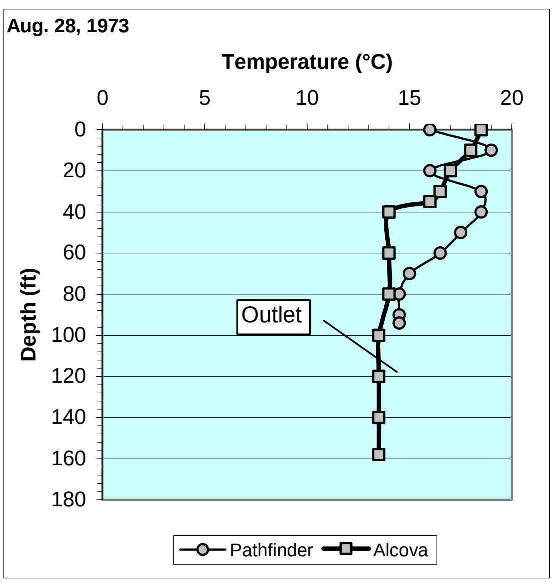


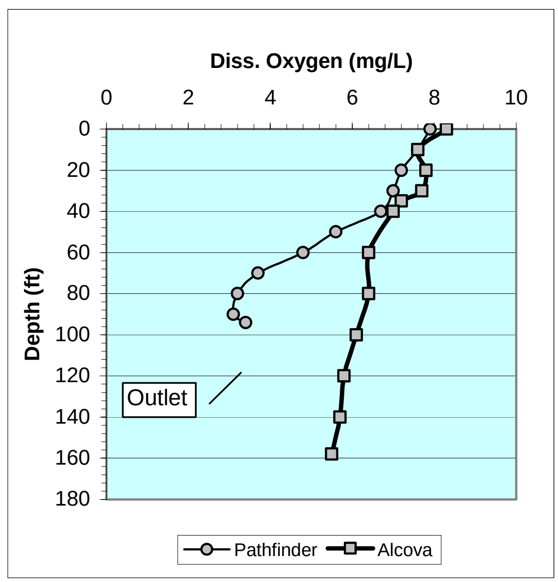
Pathfinder with a classic thermocline — well aerated throughout.
Alcova with both temperature and DO anomalies in the profiles:
near-surface temperature minimum and mid-level DO minimum.
Both minima lower than any in Pathfinder – cannot be the source.



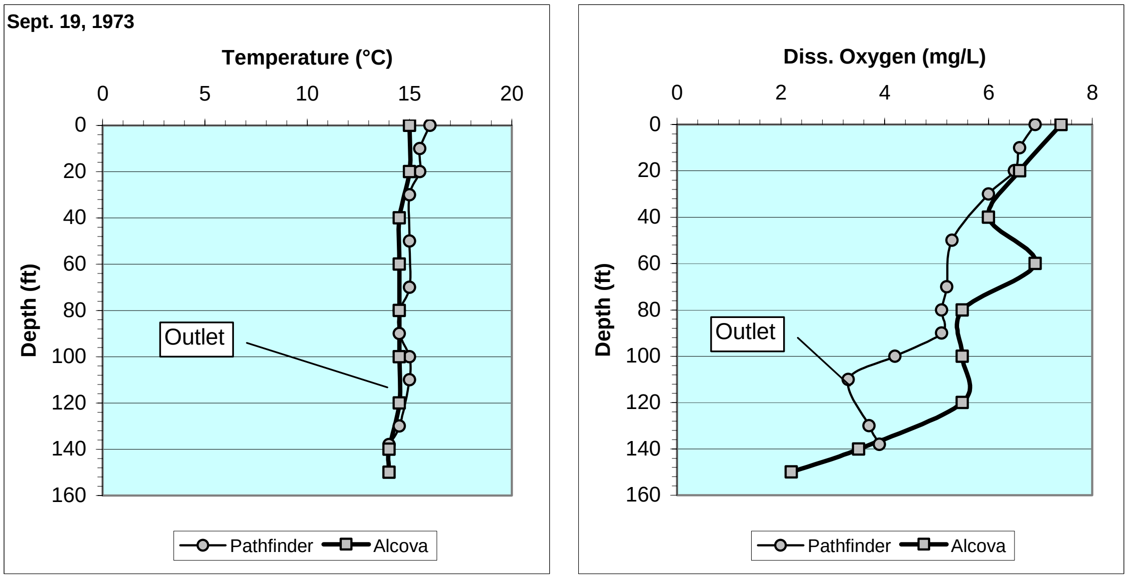


Temperature and DO profiles are similar in the 2 reservoirs.
Thermocline below the level of the Alcova outlet.
Somewhat greater DO depletion in Pathfinder.

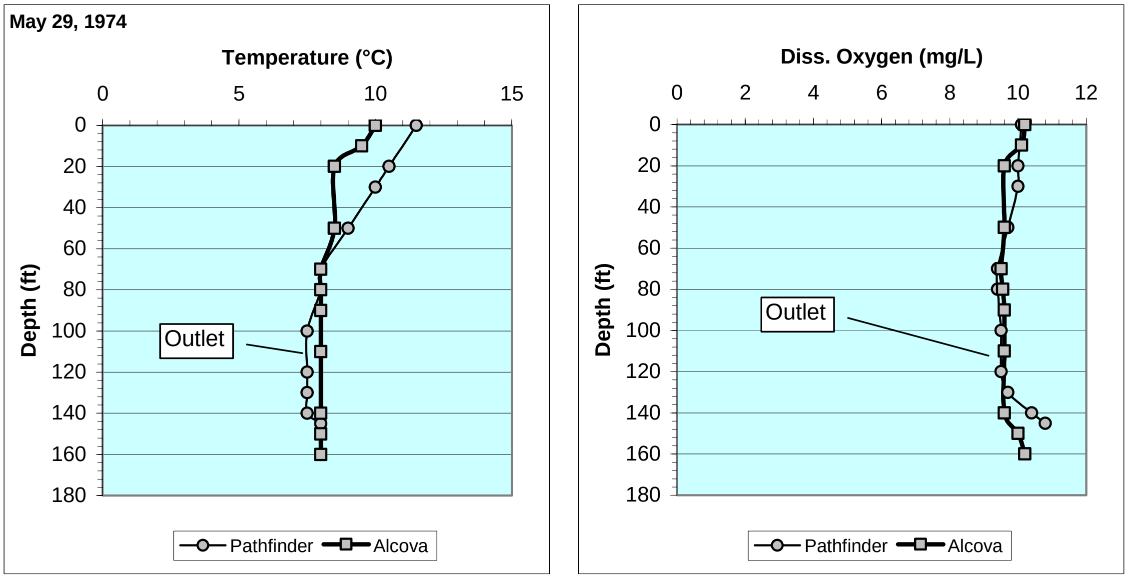




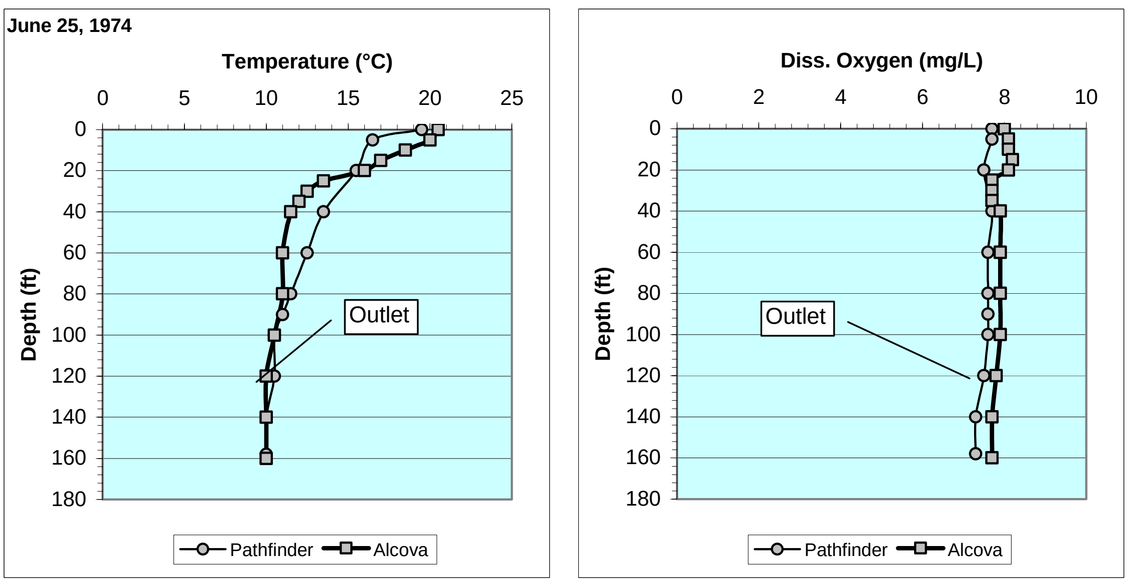
Pathfinder outlet below the bottom of the measured profiles.
Complex distribution of temperature and DO in Pathfinder Reservoir.
Alcova Reservoir strongly stratified below the level of its outlet.
No great amount of DO depletion anywhere in the Alcova profile.



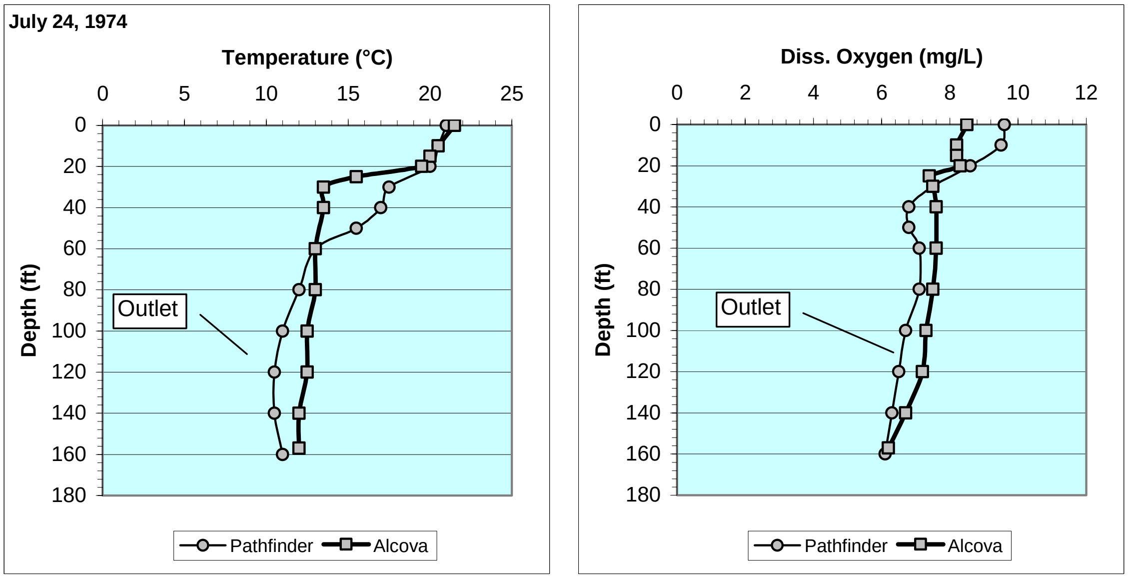
Nearly isothermal conditions in both reservoirs.
Extremely complex DO distribution in both reservoirs.
Pathfinder outlet drawing from zone of minimum DO.
Alcova profile indicates better aeration, except at the very bottom.



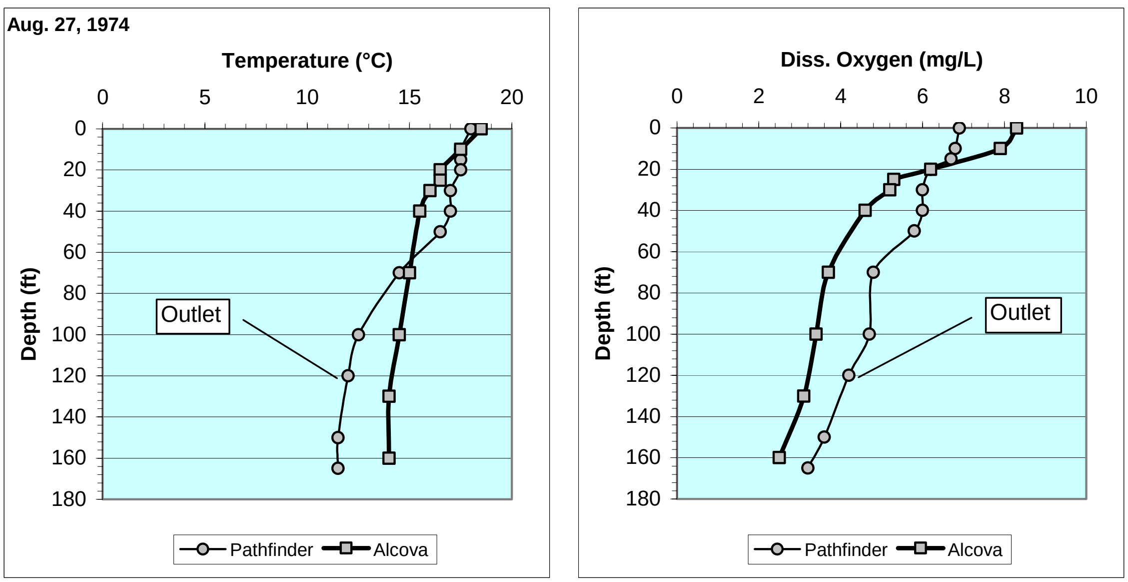
Temperatures similar at depth – Pathfinder somewhat warmer at surface.
Both reservoirs at 10± mg/L throughout the DO profile.



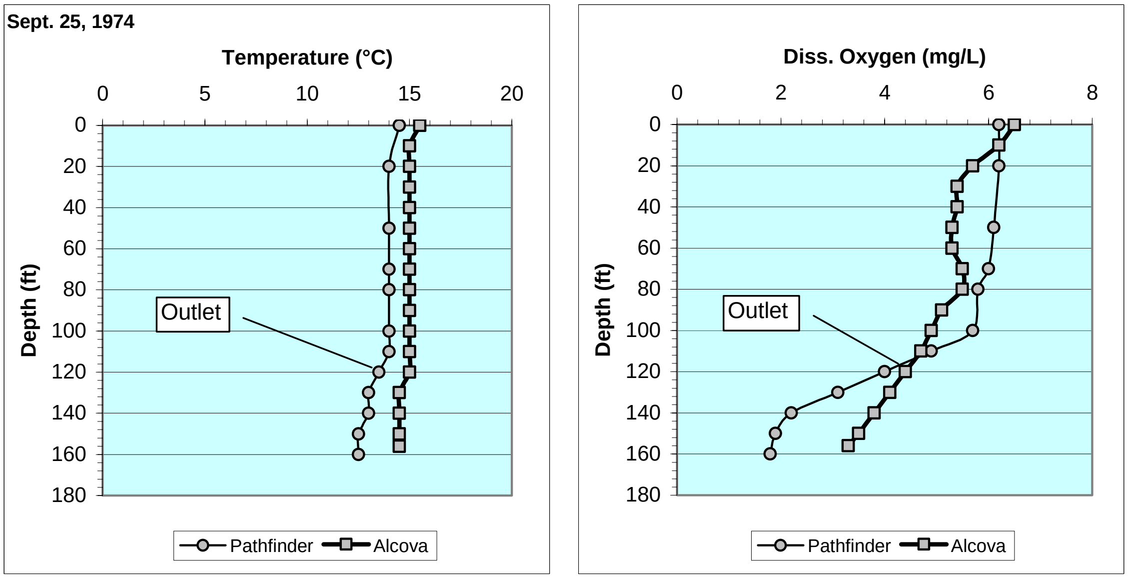
Temperature and DO distributions similar. DO at 8± mg/L in both reservoirs.



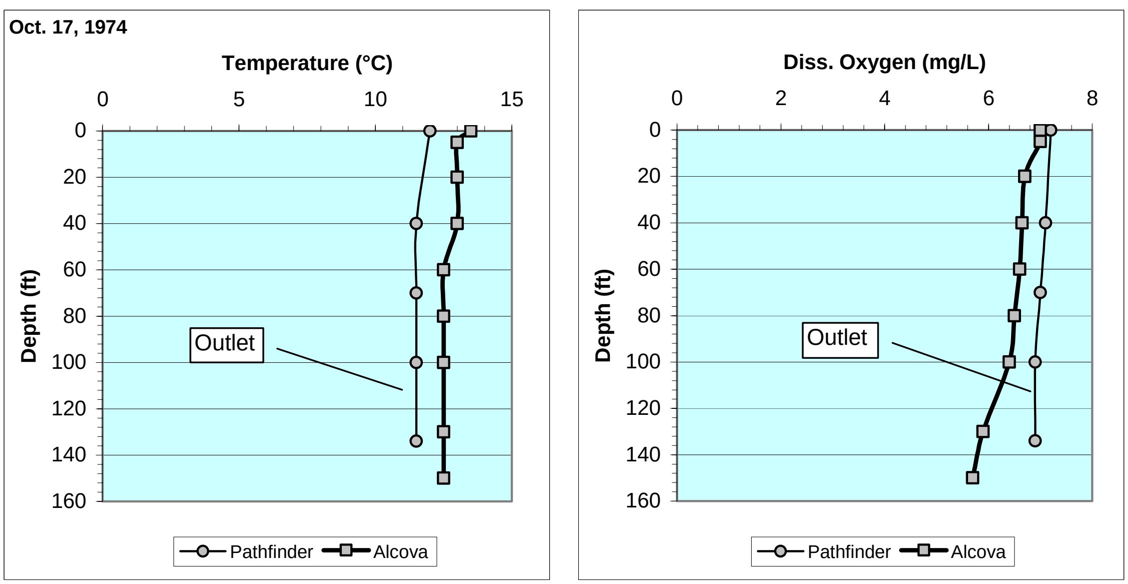
Alcova somewhat more strongly stratified, with thermocline at outlet level.
DO similar in the 2 reservoirs; slightly higher in Alcova Reservoir.



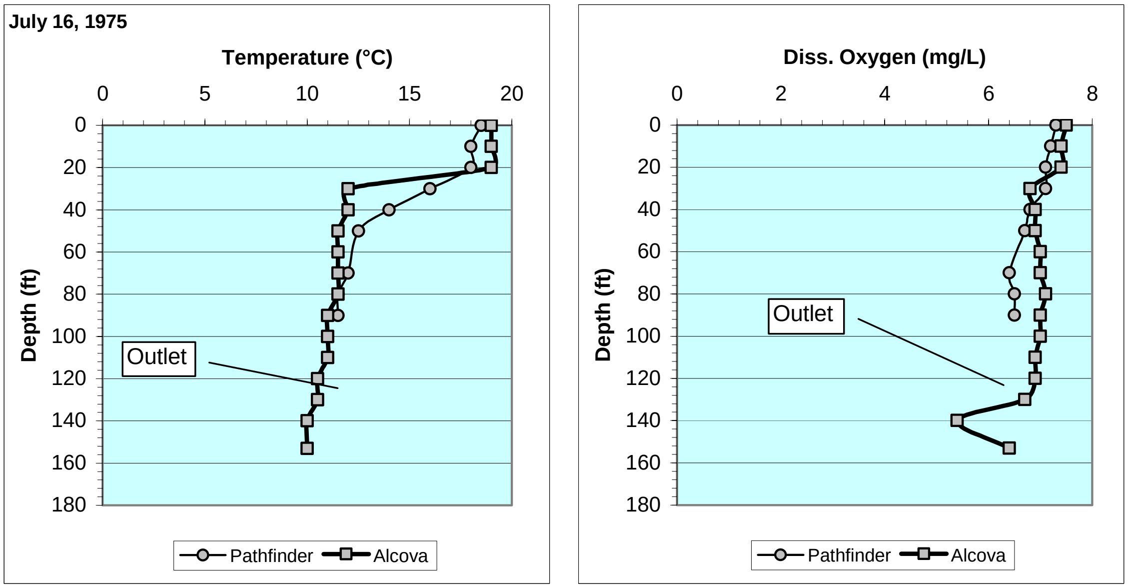
Again similar temperature profiles; Pathfinder cooler at depth.
DO lower at depth in Alcova; rapid decrease at outlet level.



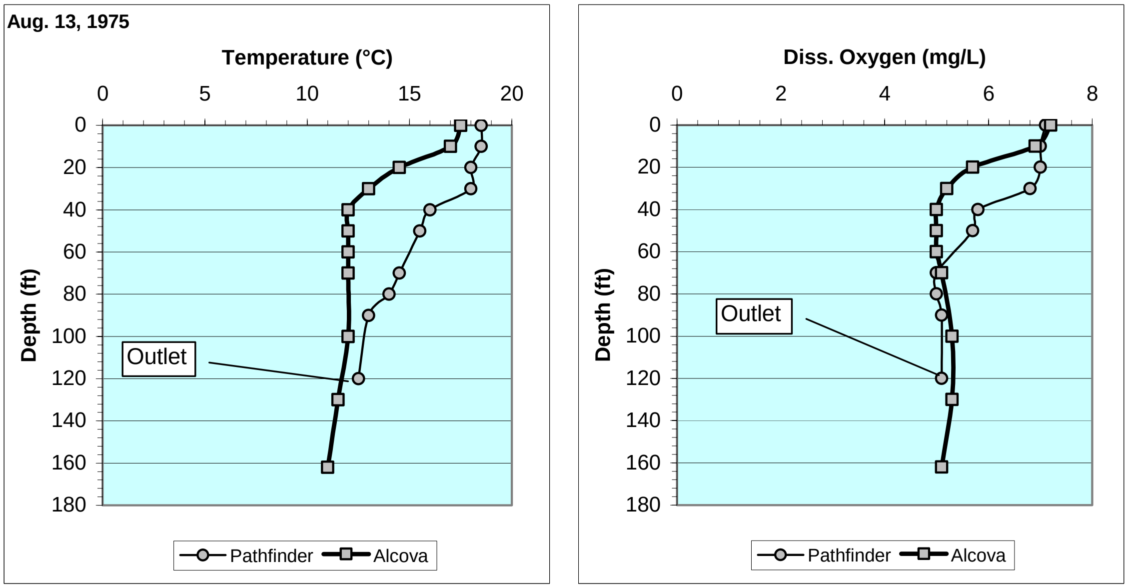
Temperature similar in the 2 reservoirs; temperature change at the level of the Pathfinder outlet in both. Decrease in DO in both reservoirs below level of temperature change. Bottom DO lower in Pathfinder Reservoir, but lower in Alcova through most of their respective profiles.



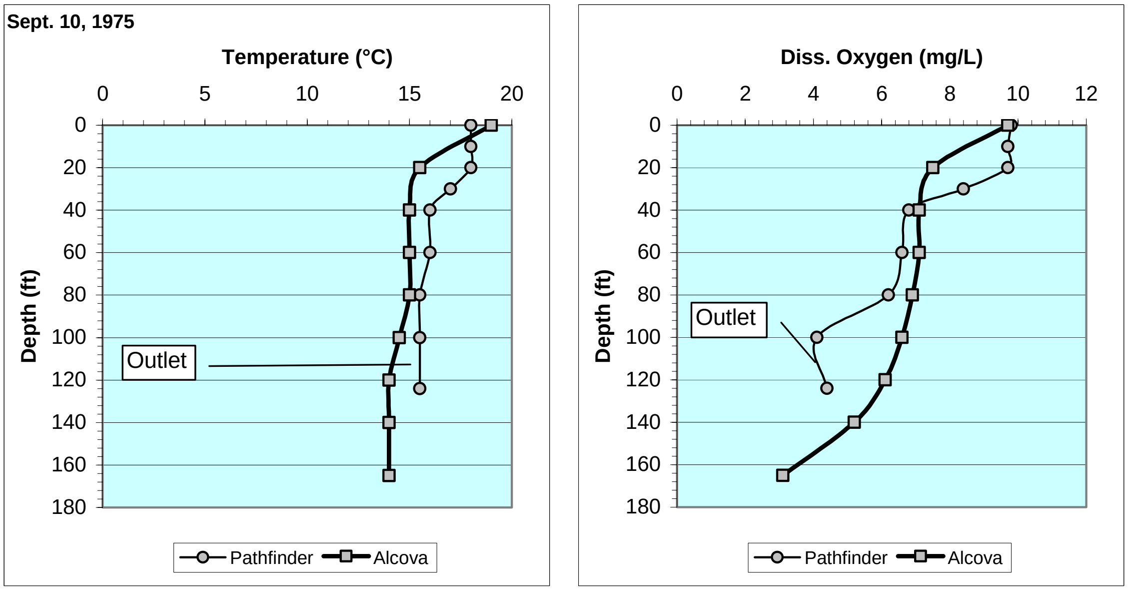
Essentially isothermal in both reservoirs. Pathfinder slightly cooler with higher DO.



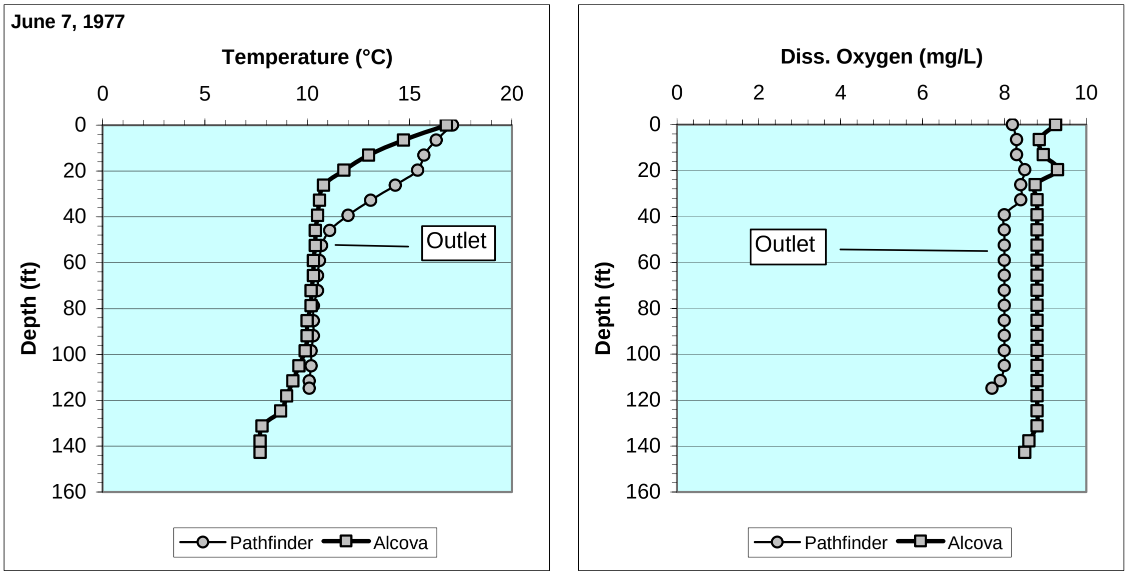
Pathfinder outlet is located below the bottom of the measured profile. Temperature profiles are similar in the 2 reservoirs. Alcova more strongly stratified with thermocline at level of outlet. DO, except at 140 feet, greater than Pathfinder DO.



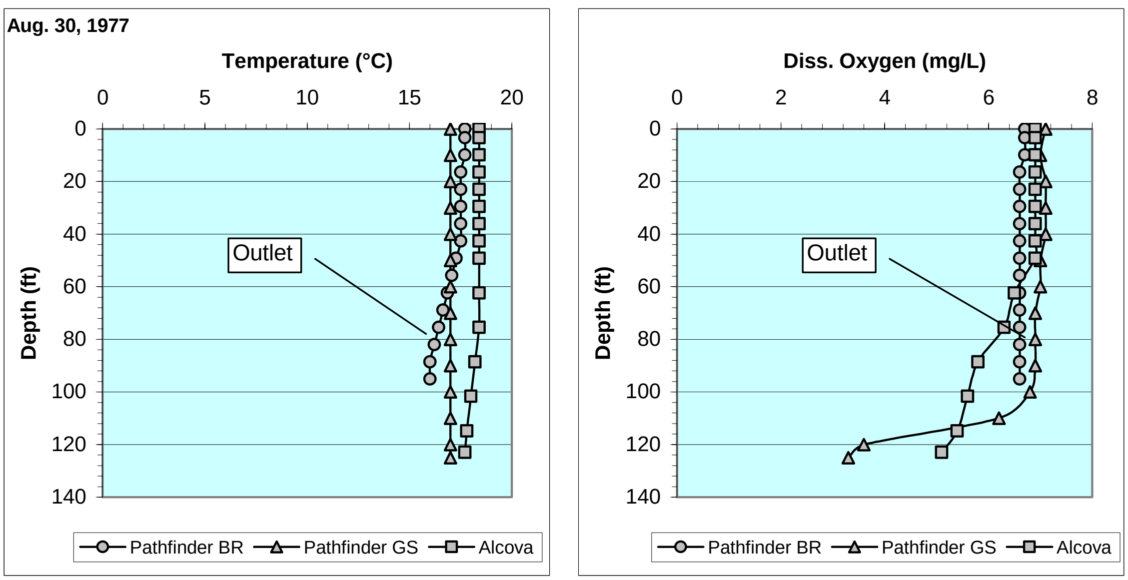
Temperature and DO profiles in the reservoirs similar to each other in form in both reservoirs. Pathfinder somewhat warmer at all levels.



A view of thermal instability in both reservoirs. Reaeration appears to be occurring, but has not completely penetrated to the bottom.

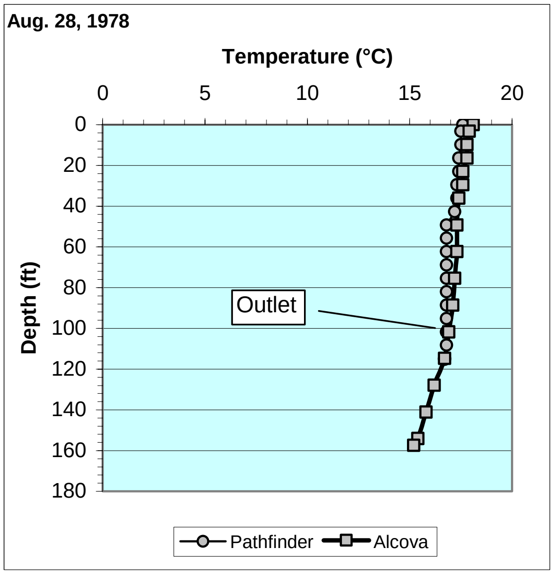


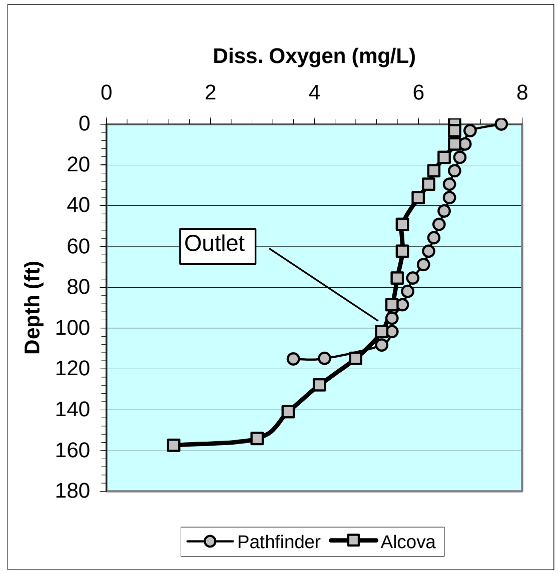
Generally similar temperature profiles – Pathfinder with a deeper thermocline.
DO profiles also similar – about 1 mg/L more DO in deeper water in Alcova.



Generally isothermal in both reservoirs. Profiles independently measured by Reclamation and USGS in Pathfinder Reservoir. DO similar in both reservoir down to about 80 feet.

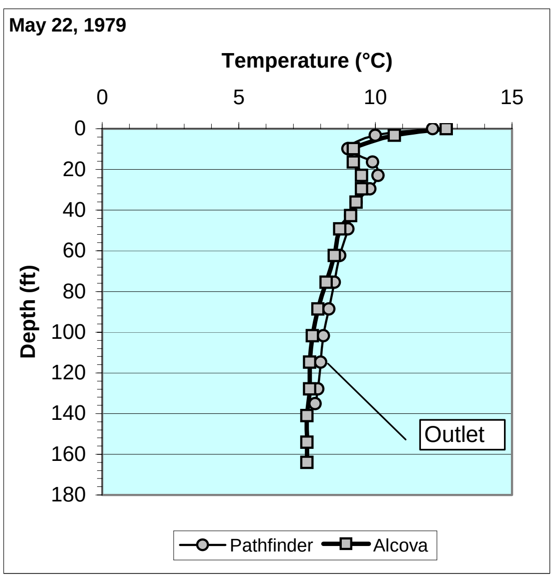
NOTE – DIFFERENT YEARS ON THIS PAGE

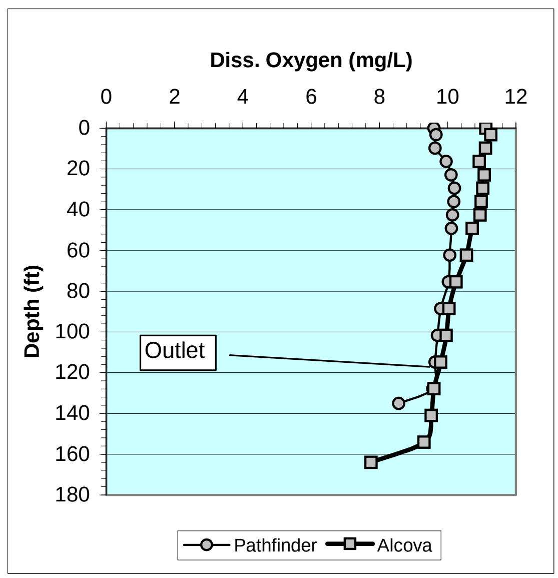




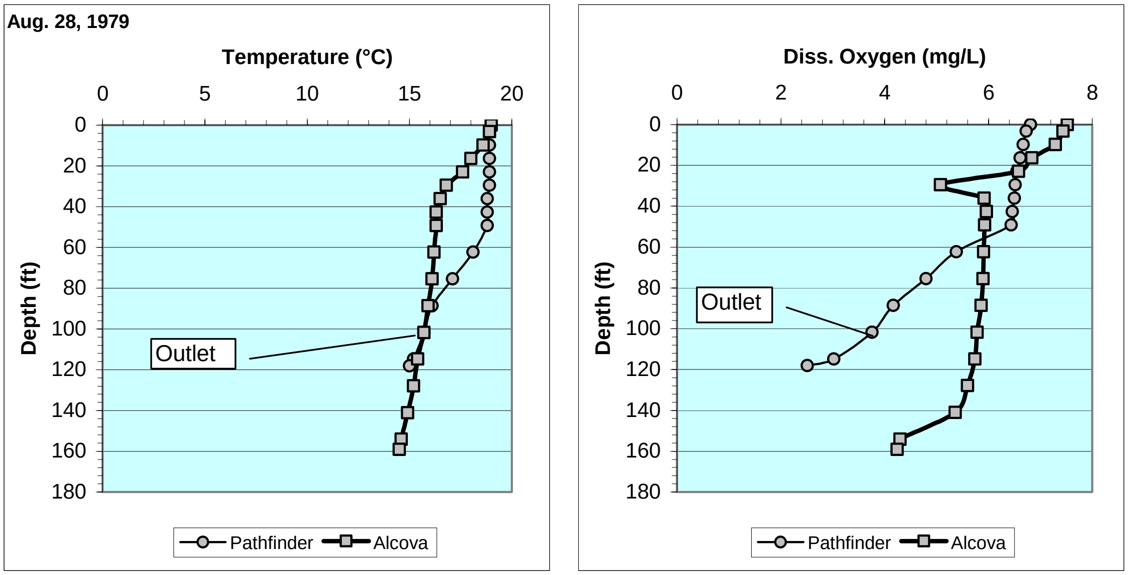
<u>These are the only profiles for 1978.</u>

Temperature profiles are essentially identical in the 2 reservoirs.
Drop off in DO below about 110 feet in both reservoirs.

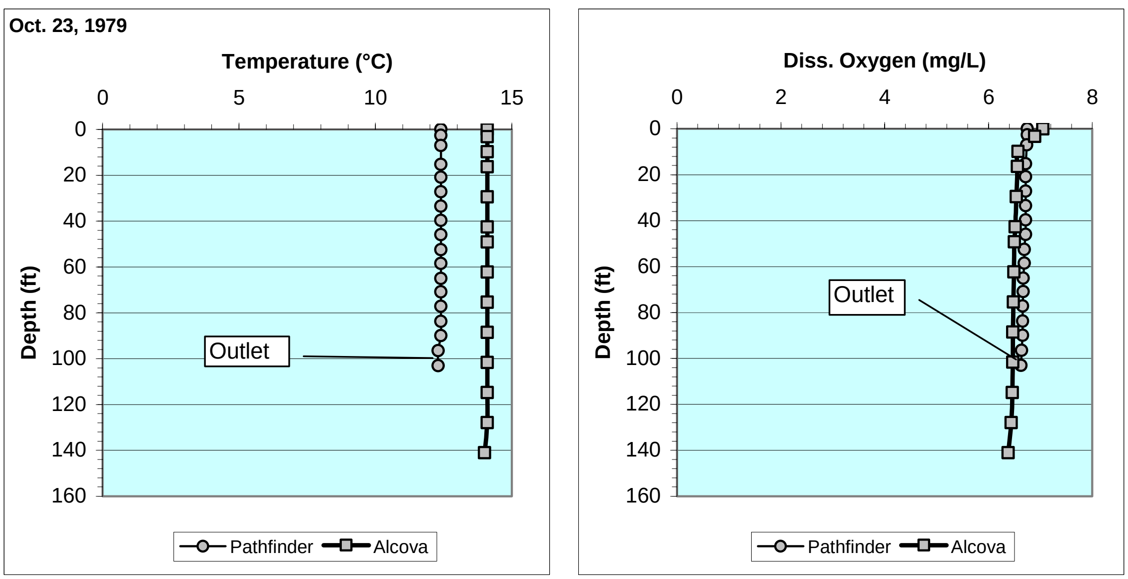




Other than surface warming, generally isothermal in both reservoirs.
DO slightly higher near the surface in Alcova, but similar at depth.



Shallower thermocline in Alcova than Pathfinder (both weak).
DO minimum below the thermocline in Alcova; dramatic decrease in DO below the thermocline in Pathfinder and continues to the bottom of the reservoir.



Both reservoirs isothermal, but Alcova is about 2°C warmer.
DO concentrations are about the same.