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Fecal Bacteria, Nutrients, Chlorophyll, and Dissolved Oxygen in a Constructed Habitat Wetland Receiving Treated Municipal Effluent and River Water

Matthew F. Knowlton and John R. Jones
Department of Fisheries and Wildlife Sciences
University of Missouri
302 Anheuser-Busch Natural Resources Building
Columbia MO 65211-7240 USA

ABSTRACT

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The Eagle Bluffs Conservation Area (Eagle Bluffs) in central Missouri includes about 400 ha of floodable pools and channels managed as seasonal or permanent habitat wetlands using treated municipal effluent from the Columbia Wastewater Treatment Wetland. Effluent, which is used year around, provides about half the annual water input and is supplemented during waterfowl migrations by pumping from the adjacent Missouri River. Infiltration rates are high ($\approx$900 cm·yr^{-1}) because of high soil permeability. Water quality of inflows and selected pools and channels on Eagle Bluffs has been monitored since wetland flooding began in 1994 because of concerns about possible negative effects of wastewater and interest in the dynamics of nutrients and algal biomass in the system. Compared to river water, effluent typically had high concentrations of fecal bacteria, chloride, dissolved phosphorus, and dissolved nitrogen, especially ammonia-N. In the water Distribution Channel (DC) and terminal wetland pools of Eagle Bluffs fecal bacteria decline rapidly and are usually within the range suitable for human water contact (<200 cells 100 mL^{-1}). Phosphorus and nitrogen concentrations vary with the relative inputs of effluent and river water and *in situ* losses. On the basis of comparison with chloride concentrations, over half of incoming phosphorus and nitrogen are lost from surface water to sediments, plant uptake or denitrification. Nutrient losses from the large proportion of water lost to infiltration are not known. Chlorophyll concentrations in the DC and wetland pools exceeded 100 µg·L^{-1} in 35% of observations, but chlorophyll and organic matter concentrations on Eagle Bluffs are similar to those in Missouri River oxbow lakes with much lower nutrient inputs. Ammonia concentration frequently exceed USEPA acute and chronic toxicity criteria, but fish seem to thrive on Eagle Bluffs. Overall, negative effects of effluent use seem minimal with respect to intended uses of the habitat wetland, but effects of infiltrating effluent on subsurface drinking water supplies are still in question.

Key Words: habitat wetland, wastewater, Missouri River, nutrients, fecal bacteria, chloride, chlorophyll, dissolved oxygen.

One means of conserving freshwater resources is to combine waters for compatible uses. An example of this practice is combining wetland wastewater treatment with creation of habitat for wildlife. Wetlands can be used very effectively for wastewater treatment (Kadlec and Knight 1996) while simultaneously providing valuable habitat (Piest and Sowls 1985, Knight 1997, Worrall et al. 1997). In most instances, however, the habitat value of treatment wetlands is a secondary consideration in their design and operation (Knight 1992). The requirements of effective wastewater treatment are sometimes at odds with those of habitat creation and wildlife in treatment wetlands sometimes create problems for system operators (Kadlec and Knight 1996, Knowlton et al. 2002). One way to maximize both the treatment performance and habitat value of treatment wetlands is to combine separately operated treatment and habitat components in the same overall system. This approach has been used in a cooperative arrangement between the City of Columbia, Missouri and the Missouri Department of Conservation (MDC) in the operation of the Columbia Wastewater Treatment Wetland (Columbia Wetland) and the Eagle Bluffs Conservation Area (Eagle Bluffs), a waterfowl management area (Brunner and Kadlec 1993). About 60,000 m^3·d^{-1} of municipal wastewater is treated by the City of Columbia and pumped from the Columbia Wetland to Eagle Bluffs where it annually provides about half the water used for wetland habitat management. Additional water comes from precipitation and the Missouri River.

Using effluent from the Columbia Wetland on Eagle Bluffs provides MDC with water they would otherwise have to pump, at considerable expense, from the Missouri River. Use of treated wastewater, however, poses some potential risks. Eagle Bluffs is a popular public use area frequented by hunters, fisherman, and birdwatchers thus making human contact with effluent inevitable. Although managed principally for waterfowl, Eagle Bluffs also supports large fish populations. The high nutrient content of wastewater could result in excessive eutrophication of the Eagle Bluffs wetlands to the detriment of fish, fisherman, and the numerous piscivores that use the area (e.g., otters, herons, eagles, cormorants). Also, infiltration of effluent into the underlying alluvium could compromise the water supply of the City of Columbia which is drawn from nearby wells. In response to these water quality concerns, MCD and the City of Columbia have sponsored long-term monitoring studies of surface and groundwater in and around Eagle Bluffs and the Columbia Wetland.

In this paper, we present results of surface water monitoring from 1994 when effluent was first used on Eagle Bluffs, through May 2001. Our principal objective is to assess whether the use of effluent in place of river water is potentially detrimental to people or wildlife using the wetlands. Possible risks include bacterial contamination and eutrophication-linked problems such as oxygen depletion, ammonia toxicity, and excessive algal blooms. Also, effluent from the Columbia Wetland annually delivers over 150 metric tons of nitrogen and 40 metric tons of phosphorus to Eagle Bluffs (Knowlton et al. 2002). Wetland wastewater treatment has a potential to reduce nutrient loading to rivers and estuaries and alter nutrient ratios that effect algal community composition (Rabalais et al. 1996). Thus our secondary objective in this paper is to describe the transformations and fate of these nutrients as they pass through this habitat wetland. We were particularly interested in whether nutrients (P, N, or SiO_2) would ever fall to potentially growth-limiting concentrations in this hypereutrophic system. In pursuit of these goals, we here present a comparison of the characteristics and dynamics of relevant water quality variables in effluent, in river water, and in the mixture of the two used in wetland management on Eagle Bluffs. We include data on fecal bacteria, plant nutrient dynamics, dissolved oxygen, and algal chlorophyll, plus background data on salinity variables useful in estimating the relative influence of different source water (e.g., effluent versus river water versus precipitation).

Study Site

The Eagle Bluffs Conservation Area is located in the Missouri River floodplain about 10 km southwest of Columbia, Missouri, in the McBaine Bottom (Fig. 1). Eagle Bluffs comprises 1683 ha of which ≈400 ha, mostly former farm land, can be flooded for wetland management (shaded areas in Fig. 1). During this study, managed wetlands on Eagle Bluffs included 13 individually controlled pools (two additional pools were impounded in 2001), a two-segment "River Supply Channel" (RSC) for distributing water from a pumping station on the Missouri River and a two-segment "Distribution Channel" (DC) for delivering river water, effluent, or a mixture to Pools 2-13. The pumping station has two, electrically operated pumps with a capacity of about 1 $m^3 \cdot s^{-1}$ each. Only river water is used in Pool 1 and the RSC to keep effluent a minimum of 1520 m from the nearest municipal water supply wells (Fig. 1). Effluent from the Columbia Wetland is aerated (cascade) then flows through a pipeline along the northern perimeters of Pools 2, 3 and 4 to a "Junction Box" – a water control structure incorporating a system of gates and diesel pumps for mixing and distributing effluent and river water and for dewatering pools when high river stages prevent gravity drainage (Brunner and Kadlec 1993). Another pipeline segment extends west from the Junction Box to a drain gate outside the main Missouri River levee (Fig. 1). Incoming effluent can be diverted from the pipeline directly into Pools 2, 3, or 4 or to the Junction Box from which it can be diverted into the Distribution Channel or directed to the drain gate. The diesel pumps allow water to be pumped to the drain gate. Water from Eagle Bluffs can also be drained from the lower end of the Distribution Channel through a gate draining into a Missouri River slough, or drained to Perche Creek via a gate at the southeast end of Pool 8 (Fig. 1).

The topography of Eagle Bluffs includes low-relief, floodplain features (channel remnants, swales) modified extensively by levees and "borrow" areas from levee construction and punctuated by several deeper scour basins ("blew holes") created by levee breaks during floods. The highest elevations (excluding artificial levees) are along the Missouri River with drainage to the south and east. Full pool surface elevations (Fig. 1) range from about 174 m above MSL for the RSC down to 172 m for Pool 11. Maximum pool depths at nominal full pool elevations range from less than 1.5 m (Pools 6, 7, 9, 11, 12 and 13) to about 2.5 m (pools 2, 3 and 8).

Roads, levees and water control structures on Eagle Bluffs sustained heavy damage during the Missouri River flood of 1993. Reconstruction was delayed by another major flood in 1995 but was largely

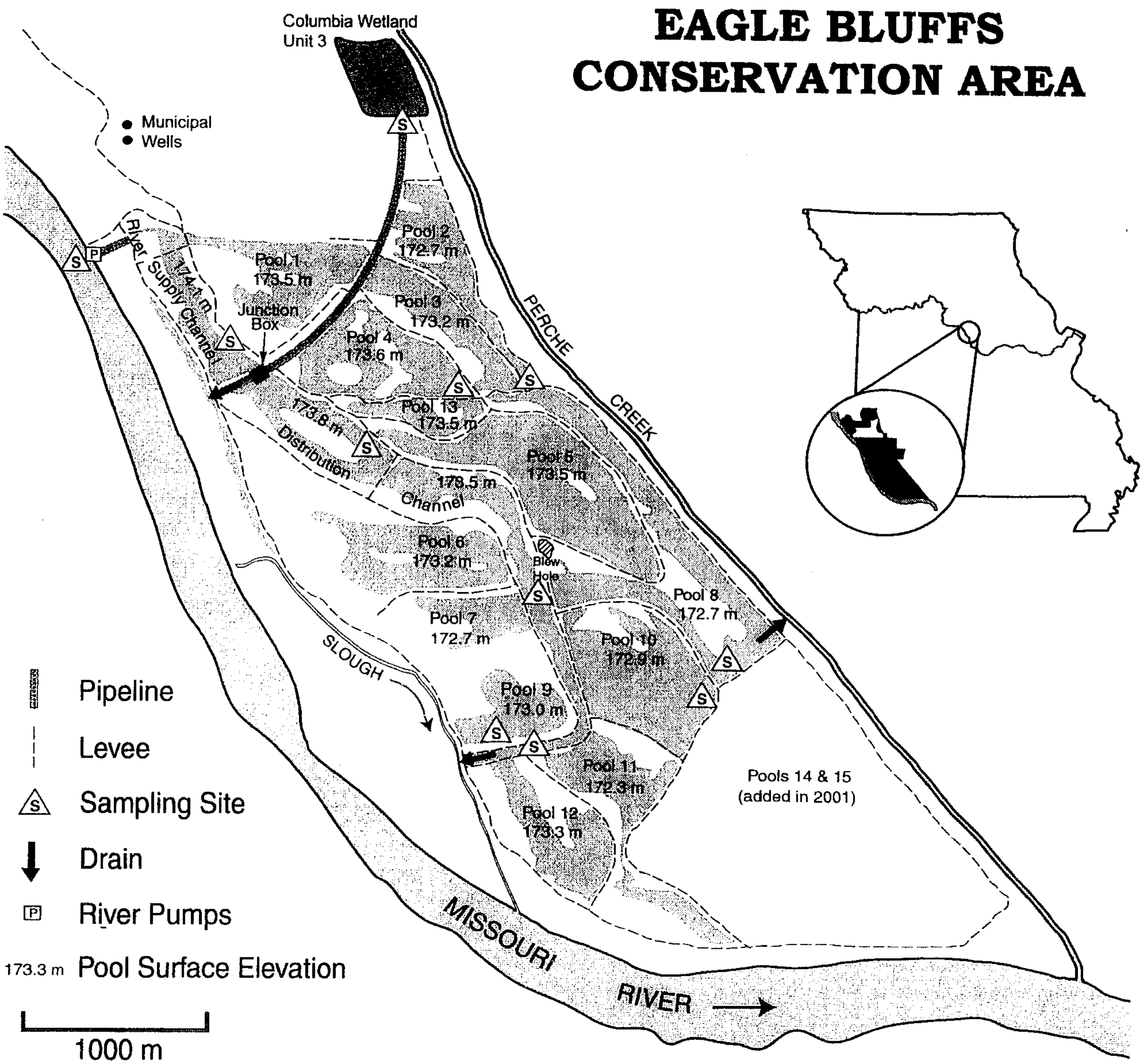

Figure 1.-Map of wetland management pools on the Eagle Bluffs Conservation Area in central Missouri. Shading represents flooded areas at "full pool" elevations.

completed by October 1995. Flooding of pools to provide habitat was begun in autumn 1994 and Eagle Bluffs was opened to waterfowl hunting in autumn of 1995.

Water Regime

In summer, portions of Pools 1-12 are usually drained, tilled and planted with grain (usually corn and soybeans) some of which is left in fields in autumn as food for waterfowl and other wildlife. Low lying areas in selected pools are kept flooded to encourage permanent marsh vegetation (mostly *Typha* sp.) or alternately flooded and drained to encourage moist-soil vegetation (dominated by species of *Carex*, *Polygonum*, and *Echinochloa*) and to provide mudflat habitat for foraging shorebirds. During waterfowl migrations in autumn and, sometimes, spring, pools are fully inundated using a combination of effluent and Missouri River water. During periods when water needs are minimal, some or all incoming effluent is drained (or, rarely, pumped) from the Junction Box to the river side of the western levee where it accumulates in borrow areas and swales extending north and south along the levee (Fig. 1). This pool sometimes overflows southward into the slough; and unused water in the lower Distribution Channel can also be drained to the slough. When full, the slough flows south to the Missouri River. Occasionally water is released to Perche Creek from Pool 8. Flow through the slough and the Perche Creek drain are not gauged, but probably amount to no more than a few percent of total inflow of effluent and river water.

In the years for which estimates are available (1996-2001), pumped inputs to the wetland portion of Eagle Bluffs (400 ha) have averaged ≈700 cm · yr^{-1} with direct precipitation and runoff contributing an additional ≈180 cm · yr^{-1}. Effluent made up about 50% of total water input during the period with river water and precipitation comprising 30% and 20%, respectively (Knowlton et al. 2002, University of Missouri unpubl. data). Pumping of river water (and total water input) usually peaked in November or December during flooding for the fall waterfowl migration. Some low-lying areas in Eagle Bluffs have relatively impervious soils and hold water, but overall, drainage is rapid. During fall (September-December) flooding periods in 1996-2001, infiltration rates averaged ≈6 cm · d^{-1} for the flooded portions of Eagle Bluffs and water residence times, calculated as total flooded volume divided by inflow rate, averaged about 10 days (University of Missouri, unpubl. data).

Materials and Methods

Sampling

Regular sampling (usually monthly) on Eagle Bluffs and the Missouri River was begun in March 1994. Routine collections have included sampling of the Missouri River, the RSC, three sites along the DC (0.8, 2.3 and 3.3 km downstream from the Junction Box), sites in Pools 2, 4, 8, 9 and 10. Sampling sites in the pools were located in low lying areas which rarely or never have cross-through flow. Hereafter we refer to these sites as "terminal pools" as distinguished from the RSC and DC where unidirectional through-flow was common. Beginning in October 1994 monthly subsamples (2 L) of outflows from the Columbia Wetland were obtained from water collected by the City of Columbia for their own monitoring purposes (Knowlton et al. 2002). Sampling for fecal bacteria was conducted monthly before 1997 and quarterly thereafter and included the sites listed above except Pools 4 and 10, which were sampled only occasionally, and a blew hole pond created by the 1993 flood at the intersection of levees separating Pool 8, Pool 5 and the DC (Fig. 1). After 1996, fecal bacteria were also collected from sites in the DC at the Junction Box and 0.1, and 0.3 km downstream.

Excluding samples provided by the City of Columbia, routine sampling involved collection of 2 L of surface water in pre-rinsed polyethylene bottles. Samples for fecal bacteria (≈50 mL) were collected in sterile polyethylene bags. Temperature and dissolved oxygen (D.O.). were measured in the field with a YSI Model 54 meter calibrated to air. Off scale D.O. measurements (>20 mg · L^{-1}) were recorded as 21 mg · L^{-1}. Samples were kept at ambient temperature in a cooler until processing (<2 hours). In ice-free periods samples were collected by wading off shore to open water locations at the sampling sites. During periods of ice cover, samples were collected by drilling through the ice with a brace and bit and extracting water using a plastic hand pump.

Analytical Methods

Unfiltered water was analyzed for specific conductance, chlorophyll (CHL)-(Knowlton 1984, Sartory and Grobbelaar 1984), total phosphorus (TP)-(Prepas and Rigler 1982), total nitrogen (TN)-(Crumpton et al. 1992), and pH (1999-2001 only). Filtrates (Whatman 934-AH glass fiber filters) were analyzed for "dissolved" total phosphorus (dTP), "dissolved" total nitrogen (dTN), nitrate-nitrite-N (NO_3-N), ammonium-N (NH_4-N)-(Stainton et al. 1977), silica, and chloride. Concentrations of particulate TP (pTP), and TN (pTN) were estimated by the difference between results for filtered and unfiltered samples. Dissolved organic nitrogen (DON) was estimated as dTN - (NO_3-N + NH_4-N). All analyses except conductivity and chloride were performed in duplicate and averaged. Unless otherwise noted, analyses were made according to A.P.H.A. (1992) or Hach (1992). Concentrations of unionized ammonia were estimated from NH_4-N, temperature and pH according to Denmead et al. (1982).

Counts of fecal coliform and fecal streptococcus were made by the membrane filtration method (U.S.E.P.A. 1978) using mFC broth with rosolic acid (fecal coliforms) and KF Streptococcus broth (fecal streptococcus).

Data Analysis

Data presented here are from October 1994 when effluent was first used on Eagle Bluffs, through May 2001. Site to site comparisons of water quality variables were made using paired tests in which differences between a pair of sites were calculated for each day sampled and a test made (two-tailed) of whether the mean difference was equal to zero. If the distribution of differences was normal (Shapiro-Wilk test), the mean was evaluated by Student's t-test (SAS 1990). Otherwise, a sign-rank test was employed. Comparisons between periods with and without river water inputs were made by one-way ANOVA (SAS 1991).

Results

Fecal Bacteria

Effluent from the Columbia Wetland is not disinfected before entering Eagle Bluffs. Both effluent and river water typically contained high numbers of fecal coliforms and fecal streptococcus. Both types of bacteria counts were usually between 10^3 to 10^4 colonies per 100 mL for effluent entering Eagle Bluffs from the Columbia Wetland (Fig. 2). Counts for the Missouri River averaged about 400 colonies per 100 mL for both bacteria types. Fecal bacteria on Eagle Bluffs declined rapidly with time as water moved through the area. Average counts declined by over a order of magnitude along the length of the Distribution Channel (Fig. 2). In the lower end of the DC, about 3.3 km downstream from the Junction Box, over 70% of coliform counts (n=28) were less than 200 colonies per 100 mL, the standard for water-contact recreation. Within 100 m of the Junction Box (n=40) only 8% of coliform counts were less than 200 per 100 mL. Counts were lower still at sampling sites in the terminal pools where water residence times are greater than in the rapidly flushed channel. Over 90% of coliform counts from pool sites (n=140) were less than the water-contact standard.

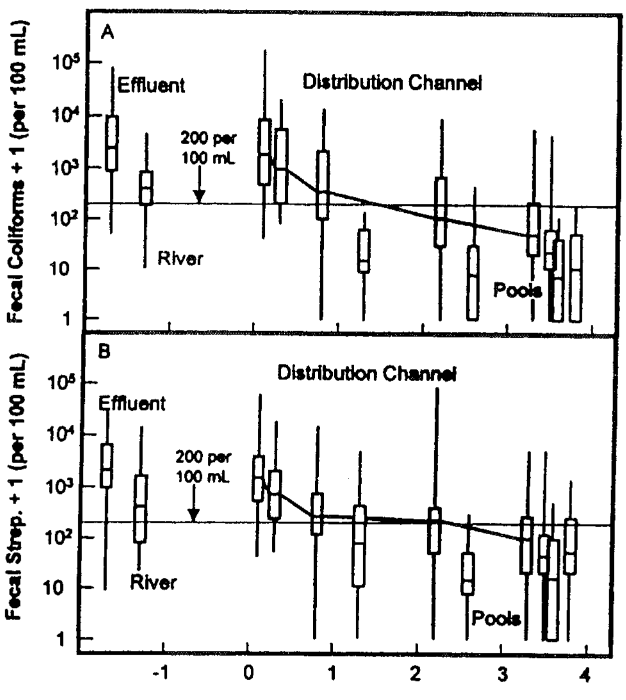

Figure 2.-Fecal coliform and fecal streptococcus concentrations from the Eagle Bluffs Conservation Area, the Missouri River and Columbia Wetland effluent. Box plots represent the mean, interquartile range, and range. The horizontal axis is the approximate distance of sampling sites from the point water enters Eagle Bluffs (Junction Box or effluent pipeline). Data from Distribution Channel sites 0 and 100 m from the Junction Box have been combined. Data for the site 2.6 km from the source are from the "blew hole" pond between the Distribution Channel and Pool 8 (Fig. 1).

Bacteria counts were quite variable at all sites, typically spanning several orders of magnitude (Fig. 2). Despite the large average difference in bacteria numbers between river water and effluent, counts on Eagle Bluffs did not differ greatly between periods with and without river water inputs. About a forth of our observations were made during periods of river pumping. Analysis of variance using counts + 1 (transformed to $\log_{10}$) showed that only a few of the sampling sites, grouped as shown in Fig. 2, showed significant effects of river water input on fecal bacteria counts of either variety. For samples collected within 100 m downstream of the Junction Box (where river water and effluent mix) coliform counts, but not streptococcus counts, did differ significantly between periods with and without river water inputs. Also, streptococcus counts, but not coliform counts, were significantly higher during periods without river water inputs for samples taken in pools approximately 3.8 km from the Junction Box. Otherwise, river inputs had no significant effect on bacteria counts.

Salinity

River water and effluent differed markedly in ionic strength and composition. Specific conductance of effluent averaged twice that of river water and had a much higher chloride content (Table 1). During periods when river water was being used for flooding on Eagle Bluffs (n=16 observations), chloride concentrations in effluent were from 8 to 22 times greater than in river water (mean=13 times greater) making chloride a useful indicator of the relative proportions of the two source waters.

Conductance and chloride concentrations on Eagle Bluffs were usually well within the range of values observed contemporaneously in the source waters (e.g., Table 1). For the most part, dynamics of salinity variables on Eagle Bluffs seems to have been a function of inflow

Table 1.–Conductance, chloride and pH data for the Missouri River, Columbia Wetland effluent, and the Distribution Channel and terminal pools on the Eagle Bluffs Conservation Area. Data cover October 1994 – May 2001. Means are shown ±1 standard deviation.

		River	Effluent	DC	Pools
Conductance	n	84	79	230	282
($\mu Si \cdot cm^{-1}$)	mean	687 ± 133	1346 ± 199	1187 ± 229	999 ± 268
	range	344 - 923	841 - 1848	721 - 1939	129 - 1794
Chloride	n	72	70	190	280
($mg \cdot L^{-1}$)	mean	19 ± 5	218 ± 47	177 ± 62	133 ± 60
	range	9 - 35	98 - 341	31 - 368	3 - 351
pH	n	27	25	81	122
	mean	8.20 ± 0.22	8.19 ± 0.11	8.31 ± 0.42	8.46 ± 0.45
	range	7.5 - 8.4	8.1 - 8.5	7.6 - 9.4	7.7 - 10.1

concentrations and mixing. Pumped inputs from the river typically diluted incoming effluent by about half as is indicated by the magnitude of seasonal declines in chloride in the upper Distribution Channel (Fig. 3). Pools being filled from the DC or the effluent pipeline generally had ion concentrations similar to the source water. Overall, ion concentrations in pools were closer to those in effluent than river water (Table 1).

Obvious effects of precipitation on ion concentrations were rare. In a few instances we did observe exceptionally low ion concentrations after heavy rains. This phenomenon was most evident during summer draw-downs when rain diluted the small volume of water remaining in a pool or refilled a previously dry pool. In drying pools we also occasionally observed increases in ion concentrations produced by evaporative concentration. For example, in July 1998 and August 1999, chloride concentration peaked in the upper DC during draw-down periods when incoming effluent was diverted to the slough (Fig. 3). In such instance, however, we never observed more than a two-fold increase in ion concentrations compared with previous samples collected during periods with fresh inflow.

pH usually increased as water flowed through the Eagle Bluffs system. In terminal pools, pH averaged almost 8.5 (Table 1) compared to 8.3 in the DC and 8.2 in both the river water and effluent. pH greater than 9 was observed in 11% of measurements in the DC and 13% of measurements in the pools.

Phosphorus

Total phosphorus in Missouri River water averaged about 17% of TP in effluent (Table 2). Phosphorus in river water was dominated by pTP which constituted an average of 60% of TP (n=84, range 33-90%). In effluent, dissolved phosphorus averaged 95% of TP (n=97, range 83-100%). Sedimentation in the River Supply Channel reduced the amount of pTP in incoming river water. During periods of pumping, we measured an average of only 26% as much pTP in the lower RSC as in river water sampled the same day (n=19, $p<0.0001$, sign-rank test). In a parallel comparison, concentrations of dTP did not consistently differ between the two sites ($p=0.41$, sign-rank test). Loss of pTP reduced TP in river inflows by an average of 42%.

Total phosphorus in the DC averaged 0.91mg $\cdot L^{-1}$ during periods with river water inputs (n=66) compared to 1.62 mg $\cdot L^{-1}$ when only effluent was used (n=174). During periods without river inputs, TP at the 0.8 km site in the DC averaged about 10% less (0.25 mg $\cdot L^{-1}$) than in effluent sampled the same day (n=56, $p<0.0001$) and TP further declined by an average of 0.54 mg $\cdot L^{-1}$

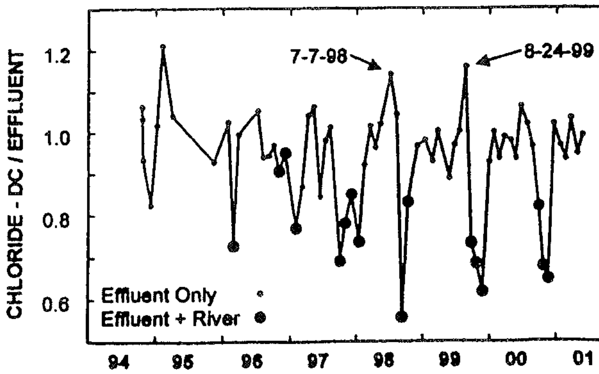

Figure 3.–Time series plot of the ratio of chloride concentrations from the 0.8 km sampling site in the Distribution Channel and effluent from the Columbia Wetland.

Table 2.–Nutrient data for the Missouri River, Columbia Wetland effluent, and the Distribution Channel and terminal pools on the Eagle Bluffs Conservation Area. Data cover October 1994 – May 2001. Means are shown ±1 standard deviation.

		River	Effluent	DC	Pools
TP ($mg \cdot L^{-1}$)	n	84	97	241	290
	mean	0.39 ± 0.31	2.21 ± 0.64	1.42 ± 0.62	0.74 ± 0.51
	range	0.12 - 1.61	0.86 - 3.96	0.35 - 3.06	0.06 - 3.75
dTP ($mg \cdot L^{-1}$)	n	84	97	244	290
	mean	0.118 ± 0.035	2.11 ± 0.63	1.23 ± 0.62	0.53 ± 0.47
	range	0.051 - 0.201	0.73 - 3.90	0.24 - 2.87	0.03 - 2.22
TN ($mg \cdot L^{-1}$)	n	84	96	241	290
	mean	2.14 ± 0.90	10.71 ± 4.56	6.70 ± 4.15	3.52 ± 3.24
	range	0.79 - 5.17	1.75 - 22.42	0.80 - 21.96	0.40 - 30.17
dTN ($mg \cdot L^{-1}$)	n	84	96	244	290
	mean	1.73 ± 0.77	10.36 ± 4.49	5.78 ± 4.43	2.37 ± 2.51
	range	0.37 - 4.07	1.49 - 21.22	0.59 - 21.55	0.36 - 13.01
NO_3-N ($mg \cdot L^{-1}$)	n	84	97	244	290
	mean	1.29 ± 0.66	1.66 ± 1.76	1.77 ± 1.56	0.71 ± 1.20
	range	0.15 - 3.44	0.03 ± 8.52	<0.005 - 8.22	<0.005 - 6.63
NH_4-N ($mg \cdot L^{-1}$)	n	84	97	244	290
	mean	0.06 ± 0.08	7.36 ± 4.29	2.91 ± 3.50	0.64 ± 1.52
	range	<0.005 - 0.36	0.15 - 18.44	<0.005 - 16.35	<0.005 - 9.82
Silica ($mg \cdot L^{-1}$)	n	73	70	191	280
	mean	12.0 ± 2.5	18.3 ± 3.5	11.8 ± 5.6	5.8 ± 5.1
	range	5.7 - 16.4	7.3 - 25.9	0.4 - 23.7	<0.1 - 24.2

(≈28%) between the 0.8 km and 3.3 km sites in the DC ($n=44$, $p<0.0001$). This change was probably not due solely to dilution. In parallel comparisons, chloride concentrations declined by an average of only 2% ($n=42$, $p<0.05$) between incoming effluent and the 0.8 km site and did not differ significantly among the three sites in the DC. Average ratios of TP:Cl^- in these samples declined significantly all along this gradient (three site-to-site pairs, sign-rank test, $n=37$ to $n=42$, $p=0.0001$ to $p=0.004$), dropping by an average of about 10% per kilometer downstream from the Junction Box. These results indicate that TP was lost, rather than merely diluted as it passed through the DC.

Losses of phosphorus in the DC seem to have affected mostly the dissolved fraction. In periods without river inputs, pTP did not consistently differ between incoming effluent and the 0.8 km site ($n=58$, $p=0.12$, sign-rank test), and tended to increase down-stream. In 73% of these observations, pTP in the DC peaked at the 2.3 km or 3.3 km sites ($n=44$). Declines in dTP, however, were typical all along this gradient. Between incoming effluent and the 0.8 km site, for example, dTP declined an average of 0.31 $mg \cdot L^{-1}$ ($n=58$, $p<0.0001$) during periods without river inputs. Between the 0.8 km and 3.3 km sites, dTP further declined by an average of 0.55 $mg \cdot L^{-1}$ ($n=44$, $p<0.0001$).

In general, phosphorus concentrations were substantially lower in terminal pools than in the source waters (DC or effluent) from which the pools were filled (Table 2). This difference results partly from the fact that effects of intermittent inputs of relatively low-P river water persist longer in the pools than in the rapidly flushed DC. But large *in situ* losses of P in the pools are also evident. For example, for pool samples with Cl^- concentration similar to those found concurrently in source waters it is likely that P concentration were also initially similar. But for pool samples with Cl^- of 90-110% of the concentration measured concurrently in source waters, TP averaged 0.58 $mg \cdot L^{-1}$ less than in the DC or effluent ($n=65$, $p<0.0001$, sign-rank test). Over 85% of these paired comparisons showed less dTP and TP in pools than source waters and a majority of the comparisons indicated >40% loss of TP in the pools. On average, pTP did not differ between pools and source waters in these samples ($n=65$, $p=0.31$, sign-rank test). Overall, TP in terminal pools averaged only 34% of TP in effluent during periods without river inputs.

Nitrogen

Like phosphorus, nitrogen was typically present in much higher concentrations in effluent than river water (Table 2). In paired comparisons, TN in the Missouri River averaged 27% of TN in effluent (n=74). Nitrogen in river water was dominated by NO_3-N (x= 58%), pTN (x=20%) and DON (x=19%). In effluent, NH_4-N averaged 65% of TN, followed by NO_3-N (x=17%) and DON (x=15%). During periods with river inputs (n=19), TN in the RSC averaged only 78% of TN in the river sampled the same day, mostly due to an average loss of 53% of pTN.

TN and NH_4-N usually declined as water passed down the DC. In paired comparisons, both variables dropped by averages of 2.1 $mg \cdot L^{-1}$ between the 0.8 km and 3.3 km sites (n=66, p<0.0001, sign-rank test for both comparisons). In periods without river inputs, TN in incoming effluent averaged 115% of TN at the 0.8 km site in the DC (mean difference = 0.95 $mg \cdot L^{-1}$, n=56, p<0.003) and 193% of TN at the 3.3 km site (mean difference = 3.20 $mg \cdot L^{-1}$, n=39, p<0.0001). For these same samples, declines in NH_4-N often exceeded those for TN (63% of observations), and averaged 1.65 $mg \cdot L^{-1}$ between effluent and the 0.8 km site (n=58, p>0.0001, sign-rank test) and 3.49 $mg \cdot L^{-1}$ between effluent and the 3.3 km site (n=39, p<0.0001).

The decline in NH_4-N in water flowing through the DC was often accompanied by an increase in the proportion of NO_3-N, suggesting nitrification of ammonia. In paired comparisons among the 0.8 km, 2.3 km and 3.3 km sampling sites in the DC, the ratio of NO_3-N to NH_4-N increased down-channel in 78% of observations (n=194), including 83% of comparisons for periods with river inputs and 75% when only effluent was entering the system. Absolute concentrations of NO_3-N, however, did not tend to increase. In paired comparisons among these three sites, there were significant differences in average NO_3-N only between the 2.3 km and 3.3 km sites and NO3-N showed an average net decline, down-channel over that distance (n=66, p=0.0022, sign-rank test, mean difference = -0.19 $mg \cdot L^{-1}$). Average DON did not differ significantly among the three sites.

In most observations, total and inorganic nitrogen occurred in terminal pools at lower concentrations than in source waters. These differences were probably due to loss of N rather than dilution. For paired observations in which Cl in pools was 90-110% of Cl in inflowing water (DC or effluent pipeline), TN, NO_3-N, NH_4-N, respectively averaged 3.13 $mg \cdot L^{-1}$ (n=64, p<0.0001, sign-rank test), 0.97 $mg \cdot L^{-1}$ (n=65, p<0.0001, sign-rank test) and 2.42 $mg \cdot L^{-1}$ (n=65, p<0.0001, sign-rank test) less in pools than source waters. For TN, this difference represents an average decline of almost 40%. DON, on the other hand, did not consistently differ between pools and source water in these comparisons (n=65, p=0.37, sign-rank test). In the entire data set, NO_3-N was undetectable in pools (<0.005 $mg \cdot L^{-1}$) in about a fourth of the observations (n=76/290) and NH_4-N was below detection (0.005 $mg \cdot L^{-1}$) in about a fifth of observations (n=52/290). NO_3-N and NH_4-N in pools were less 0.05 $mg \cdot L^{-1}$ in 46% and 53% of the measurements, respectively. Overall, TN in terminal pools averaged only 36% of TN in effluent during periods without river inputs.

Concentrations of NH_4-N in the DC and terminal pools were negatively correlated with pH (r=-0.45, n=203, p<0.0001) and water temperature (r=-0.36, n=528, p<0.0001). Thus equilibrium conditions did not favor occurrence of ammonia in its unionized form. Estimated concentrations of unionized ammonia never exceeded 0.5 $mg \cdot L^{-1}$ (n=201) and only 3% of observations exceeded 0.2 $mg \cdot L^{-1}$.

Silica

In paired samples, silica in the Missouri River (Table 2) averaged 69% of silica in effluent (n=66). In the Distribution Channel silica usually declined downchannel. In paired observations, the 0.8 km and 3.3 km sites differed by an average of 3.8 $mg \cdot L^{-1}$ or about 23% (n=53, p<0.0001, sign-rank test). During periods without river inputs, silica concentrations at the 0.8 km site averaged 3.23 $mg \cdot L^{-1}$ (about 20%) less than in the incoming effluent (n=43, p<0.0001, sign-rank test). Silica declined further in the pools. For observations in which Cl in the pools was 90-110% of that in source waters, silica averaged 5.4 $mg \cdot L^{-1}$ less in the pools (n=65, p<0.0001), a typical decline of > 45%. Among all data from the terminal pools (n=280) silica was below 1 $mg \cdot L^{-1}$ in about 10% of observations and below 0.5 $mg \cdot L^{-1}$ in 2.5% of the observations. Overall, silica in terminal pools averaged only 29% of silica in effluent during periods without river inputs.

Chlorophyll

Median CHL concentrations in the river and effluent were 21 $\mu g \cdot L^{-1}$ and 12 $\mu g \cdot L^{-1}$, respectively. CHL in the river varied seasonally, usually peaking in spring or summer with a minimum in winter (Knowlton and Jones 2000). In effluent, CHL declined over time as vegetation (mostly *Typha latifolia*) in the Columbia Wetland matured (Knowlton et al. 2002). Both source

Table 3.–Chlorophyll and dissolved oxygen data for the Missouri River, Columbia Wetland effluent, and the Distribution Channel and terminal pools on the Eagle Bluffs Conservation Area. Data cover October 1994 – May 2001. Means are shown ±1 standard deviation.

		River	Effluent	DC	Pools
CHL ($\mu g \cdot L^{-1}$)	n	83	91	242	285
	mean	28 ± 22	21 ± 30	102 ± 158	146 ± 266
	range	4 - 107	3 - 239	3 - 847	<1 - 3296
D.O. (% saturation)	n	86	24	246	290
	mean	87 ± 11	85 ± 15	82 ± 31[1]	91 ± 36[2]
	range	48 - 122	66 - 139	9 - 27	7 - 240

[1] Data include 2 observations with D.O. >20 $mg \cdot L^{-1}$ that were recorded as 21 $mg \cdot L^{-1}$.

[2] Data include 11 observations with D.O. >20 $mg \cdot L^{-1}$ that were recorded as 21 $mg \cdot L^{-1}$.

waters exhibited a wide range of CHL with peak concentrations over 100 $\mu g \cdot L^{-1}$ (Table 3).

Incoming river water usually lost CHL in transit. In paired comparisons for periods with river inputs, CHL was always less in the lower RSC than in the river sampled on the same day (n=19, median difference = 10 $\mu g \cdot L^{-1}$, $p<0.0001$ – rank-sign test). In contrast, CHL in incoming effluent often increased after entering the DC. In periods without river inputs, average CHL in the upper DC (0.8 km site) was significantly greater than in incoming effluent (n=57, p=0.0049, sign-rank test). Along the DC, CHL increased by a median amount of 16 $\mu g \cdot L^{-1}$ between the 0.8 km and 3.3 km sites (n=65, $p<0.0001$ – sign-rank test). And CHL was usually greater in terminal pools than in the waters feeding them. In observations where Cl^- in pools was 90-110% of Cl^- in their source waters, CHL in pools was higher by a median amount of 14 $\mu g \cdot L^{-1}$ (n=65, $p<0.015$ – sign-rank test). CHL at all nine Eagle Bluffs sampling sites was negatively correlated to water depth (n= 41 to n=94, r= -0.25 to r=-0.61) although at one site the correlation was significant only at the 6% level. CHL was not very seasonal, algal blooms with CHL >300 $\mu g \cdot L^{-1}$ were observed in every month of the year. CHL, however, was significantly positively correlated to water temperature at four of the nine sites (r=0.36 to r=0.44).

In general, the DC and pools of Eagle Bluffs supported high concentrations of algal biomass (Table 3). Collectively, 35% of samples from these sites had CHL over 100 $\mu g \cdot L^{-1}$ and 8% had CHL over 400 $\mu g \cdot L^{-1}$. Variation in algal biomass also largely controlled the dynamics of particulate nutrients. Excluding a few influential observations with CHL over 1000 $\mu g \cdot L^{-1}$ (n=4), regression analysis shows variation in CHL to account for 73% of the variation in pTP (n=505, $p<0.0001$, MSE = 0.012) and 79% of the variation in pTN (n=490, $p<0.0001$, MSE = 0.223).

Dissolved Oxygen

Water in the Missouri River, incoming effluent, and in pools and channels on Eagle Bluffs tended to be undersaturated with dissolved oxygen (Table 3). Oxygen in the DC averaged 84% of saturation during periods without river water inputs (n=176) compared to 78% when source waters were combined (n=69) and D.O. usually increased as water moved down the channel. In paired comparisons, D.O. averaged about 1.5 $mg \cdot L^{-1}$ more at the 3.3 km site than the 0.8 km (n=66, $p<0.0001$, sign rank test) and 2.3 km (n=66, $p<0.0001$, sign rank test) sites. The latter two sites did not significantly differ (n=68, p=0.19, sign rank test). Oxygen was higher still in terminal pools which averaged 1.5 $mg \cdot L^{-1}$ more D.O. than the average of DC sites sampled on the same day (n=76, $p<0.0001$, sign rank test). But despite this downstream gradient of increasing D.O., over 60% of measurements in terminal pools were below saturation.

Effluent enters Eagle Bluffs with an unsatisfied BOD averaging about 8 $mg \cdot L^{-1}$ (Knowlton et al. 2002). Also, the oxygen data contain a negative sampling bias because collections were usually done in the morning except during waterfowl hunting seasons. About 78% of oxygen measurements were taken less than 5 hours after sunrise (median = 3.1 hours after sunrise). Nonetheless, oxygen supersaturation was not uncommon. DO above 110% saturation was observed in about 13% of measurements in the DC and 27% of measurements in terminal pools. And severely low D.O. was relatively rare. Only 15% of pool and channel measurements were below 5 $mg \cdot L^{-1}$ and less than 3% were below 3 $mg \cdot L^{-1}$. Low D.O. was most frequent during summer draw-downs which sometimes caused fish-kills in the shrinking pools. Low D.O. was also observed during sample collections. Individuals most often observed are "rough" fish such, carp (Cyprinus and Hypophthalmichthys) and gar (Lepisosteidae) which may be less sensitive to a degraded environment than other groups, but are typical inhabitants of riverine wetlands (Pflieger 1997). It seems unlikely that ammonia in effluent has impaired the ability of Eagle Bluffs to provide fish for anglers and fish-eating wildlife.

Incoming effluent is only moderately saline with about twice the ionic strength of river water (Table 1) and evaporative concentration of salts has not been observed to increase dissolved solids much above the range seen in normal effluent (e.g., Fig. 3). In central Missouri, evaporation and precipitation roughly balance (Missouri Department of Natural Resources 1986) and with the high infiltration rates of soils in the Eagle Bluffs wetlands, there is little likelihood of extensive salt build up in soils. Nonetheless, relatively small increases in soil salinity could have important effects. Greenhouse studies using Eagle Bluffs soils watered with deionized water, effluent, or river water showed an unexplained impairment by effluent of the growth, biomass and species diversity of native plants (Finnocchiaro 2000). Effluent-treated soils also ended up with about twice the sodium concentration measured in other treatments and this difference may have contributed to the poor growth of the plants. If these results are confirmed in the field it may suggest the need for managing water inputs to minimize soil sodium concentrations by providing a flush of low-sodium river water prior to seasonal draw-downs (Finocchiaro 2000).

Algal blooms are a frequent phenomenon on Eagle Bluffs where chlorophyll concentrations in the DC and terminal pools are often over 10 times the 15 $\mu g \cdot L^{-1}$ average found in a Missouri-wide survey of reservoirs (Jones and Knowlton 1993). With high nutrient loading from both river water and effluent, it is unlikely that algae in Eagle Bluffs waters are nutrient limited. Phosphorus limitation is particularly unlikely. Dissolved P rarely drops below 50 $\mu g \cdot L^{-1}$ (2 of 534 observations in the DC and pools) with the majority of observations above 700 $\mu g \cdot L^{-1}$. Occasional measurements (n=30) have shown dTP to be mostly in the soluble reactive fraction ($\bar{x}$=72%). Less than 5 $\mu g \cdot L^{-1}$ of SRP is usually sufficient to prevent P-limitation of algal growth (Reynolds 1992). Nitrogen may occasionally be limiting as dissolved inorganic nitrogen (NO_3-N + NH_4-N) falls into a potentially growth-limiting range (<100 $\mu g \cdot L^{-1}$ – Reynolds 1998) in about a fourth of our observations, but those samples (n=137) contained an average of 1.0 $mg \cdot L^{-1}$ of DON and preliminary lab studies (University of Missouri, unpubl. data) have shown this fraction to be at least partly available for algal growth. Silica, too, seems usually to be present in excess quantities. Fewer than 2% of our measurements in the DC and pools had silica less than 0.5 $mg \cdot L^{-1}$ and thus, diatom growth was probably not inhibited by silica depletion (Reynolds 1984). Excess nutrient concentrations, together with shallow water depths and the suppression of macrophytes by fish make Eagle Bluffs an excellent environment for algal growth.

Further evidence of the apparent lack of nutrient limitation of algae on Eagle Bluffs can be found in a comparison of data between Eagle Bluffs and other, less nutrient-rich waterbodies. In 1994-1998 we studied four shallow oxbow lakes in the Missouri River floodplain in west-central Missouri (Knowlton and Jones 1997) as part of a multi-disciplinary habitat evaluation in the aftermath of the 1993 Missouri River flood (Galat et al. 1998). Collection procedures and sample analysis were nearly identical to the current study. As a group, the oxbows exhibited much lower concentrations of dissolved N and P than usually occur on Eagle Bluffs (Table 4), but supported similar average concentrations of CHL. The proportion of samples with CHL over 100 $\mu g \cdot L^{-1}$ was actually higher in the oxbows (47%) than in the DC (20%) or terminal pools (40%) suggesting that the frequency of large algal blooms was not increased by the higher nutrient loading in the Eagle Bluffs system. Thus it is not obvious that use of effluent on Eagle Bluffs has resulted in more or larger algal blooms than would occur if only river water was used in the system.

Oxygen depletion in Eagle Bluffs wetlands does not seem to have been a problem insofar as acutely low D.O. levels has been relatively rare. Only 3% of D.O. measurements were below 3 $mg \cdot L^{-1}$ despite the preponderance of early-morning observations in the data set. Oxygenated refugia for fish were always present except in areas intentionally drained for other purposes. High rates of algal photosynthesis and atmospheric inputs seem usually to more than offset the unsatisfied oxygen demand of incoming effluent and the respiratory needs of resident biota. River water, having lost much of its suspended load in transit through the RSC, may represent a lesser source of oxygen demand than wastewater. Thus, given the lack of clear evidence that effluent has a positive effect on algal productivity, it is arguable that oxygen conditions would be at least somewhat better without inputs of effluent. Nonetheless, current conditions seem adequate for maintenance of fish populations.

Overall, the data from this study suggest that effluent has no clearly demonstrated negative effects on the Eagle Bluffs system that would negate its virtue of being a source of low-cost water for wetland management. Use of effluent on Eagle Bluffs is controversial, nonetheless, because of the proximity of municipal water supply wells (e.g., Fig. 1). Management quite rare during fall flooding, despite the inundation of hundreds of hectares of partially harvested field crops and other dead or senescent vegetation. We observed only 7 instances in which D.O. in vegetated terminal pools dropped below 5 mg · L^{-1} during fall flooding (n=95). This last result may also reflect a positive sampling bias because our collections were made in deeper, uncultivated areas some distance from the heaviest vegetation and litter because such areas would constitute refuges for oxygen-stressed fish. We also saw little depletion of dissolved oxygen under ice cover. Of the 65 oxygen measurements made in pools or channels under complete ice cover only 3 were below 50% saturation (mean=96% saturation, range 28-164%).

Discussion

Surface water monitoring of the Eagle Bluffs wetlands has been conducted to address the general question of whether use of effluent creates problems with respect to the management of the system. Potential problems include anthropogenic toxicants, microbiological contamination of visitors and wildlife, ammonia toxicity, algal blooms, and oxygen depletion. Based on the accumulating data from other treatment wetlands (e.g., Kadlec and Knight 1996, U.S.E.P.A. 1999a) it was expected that such problems, if they occurred, would not be severe and monitoring data to date seem to bear out that expectation.

If effluent were unavailable, wetlands on Eagle Bluffs would be maintained using water from the Missouri River. The Missouri River transports considerable quantities numerous anthropogenic chemicals (Goolsby and Battaglin 1993) that have resulted in official advisories against eating certain river fishes (e.g., PCB contamination of sturgeon, Gale Carson, Missouri Department of Health, news release, 6 July 2001). The effluent stream treated by the City of Columbia, in contrast, includes no major industrial waste and is largely free of heavy metals and potentially toxic organic materials. Neither water source is ideal. River water is more turbid than effluent. River water is also nutrient-rich and carries high concentrations of fecal bacteria, though less so on both counts than the wastewater and the RSC acts as a settling basin removing a substantial proportion of riverine seston, including a large fraction of the particulate N and P and chlorophyll, before river water enters the main wetland complex.

With respect to bacteria, counts of fecal coliforms and fecal streptococcus on Eagle Bluffs tend to be much less than in the Missouri River except in the immediate vicinity of the effluent outfall (Fig. 2). Coliform counts in terminal pools, in fact, usually meet the standard for whole-body water contact recreation, a very conservative standard given that most human activities on Eagle Bluffs (e.g., hunting, fishing, bird-watching) involve minimal water contact. Fish growing in effluent may harbor human pathogens that might pose a threat to people handling and consuming contaminated individuals (Janssen 1970), but examples of fish acting as vectors of infectious disease are rare (Tsai 1975). Likewise, the likelihood of pathogens in effluent harming fish and wildlife in the wetlands seems remote (Friend 1985, Brennan 1985). Overall, effluent may pose little, if any, greater risk than river water as a source of infectious or toxic contaminates with the exception of ammonia.

Concentrations of NH_4-N are typically quite low in the Missouri River and quite high in effluent (Table 2). On Eagle Bluffs, NH_4-N usually declines rapidly downstream from the effluent outfall and is negatively correlated to pH and temperature. Thus concentrations of toxic, unionized ammonia are relatively low. The growth of algae and macrophytes that produce the elevated pH responsible for increased proportions of molecular ammonia also create a large biotic demand for inorganic nitrogen that serves to decrease NH_4-N concentrations generally through plant uptake and nitrification. Nonetheless, NH_4-N at Eagle Bluffs frequently exceeds the USEPA acute and chronic criteria for ammonia which are based on NH_4-N concentration, pH and temperature (U.S.E.P.A. 1999b). The USEPA acute ammonia criterion is a pH adjusted NH_4-N concentration that, it is recommended, should be exceeded no more than once in three years. But in 1999-2001, the acute ammonia criterion (for waters without Salmonids) was exceeded in 30% of our observations in the Distribution Channel and 9% of our observations from terminal pools. These proportions would likely be even higher if samples had not routinely been taken in the morning before the peak of photosynthetic activity. The USEPA chronic ammonia criterion for waters with "fish early life stages present" was exceeded in 63% of observations in the DC and 15% of observation in pools. Our data are one-day measurements rather than the 30 day average concentrations recommended for evaluating the chronic criterion (U.S.E.P.A. 1999b), but it is doubtful that more frequent sampling would reveal any lesser degree of impairment.

No formal studies have yet been made of aquatic invertebrates or fish on Eagle Bluffs, thus any negative effect of ammonia toxicity cannot be directly evaluated. Fish do occur in large numbers in all perennially flooded channels and pools on Eagle Bluffs despite occasional die-offs caused by summer draw-downs. On-site reproduction is likely as juvenile fish are often flooding has raised the surface of the water table under Eagle Bluffs so that it is now sometime over 3 m higher than at the nearest municipal wells. This situation creates a strong possibility that water applied to the surface at Eagle Bluffs with eventually migrate north into the well field alluvium. If this occurs, it is likely the city water supply will be officially reclassified as "groundwater under direct influence of surface water," a status that carries requirements for much more rigorous treatment and testing (Missouri Department of Natural Resources 2000) regardless of whether river water or effluent is the surface water source. Given the level of treatment effluent receives in the Columbia wastewater treatment plant and wetlands (Knowlton et al. 2002) coupled with the additional "polishing" and dilution provided by the Eagle Bluffs system and transit through the alluvium, it is doubtful that interception of some effluent-amended groundwater would pose much threat to the Columbia drinking water supply. But news of incursions of effluent into the drinking water supply might be poorly received by the public and a downgrading of water supply status would require expensive modifications of the Columbia water treatment system. Thus the overall cost to the public of the Eagle Bluffs operations may not be known for some time.

The dynamics of phosphorus and nitrogen in Eagle Bluffs are consistent with expectations for this highly productive system (Howard-Williams 1985). The precipitous declines in dTP and DIN "downstream" as water moves from inflows to the terminal pools reflect uptake by algae and macrophytes and the ultimate loss of nutrients to the sediments or denitrification. Comparisons of average TP and TN between effluent and terminal pools suggest removal of over 60% of both nutrients, but actual net removal rates may be greater or less. Tracking specific water masses through the area is difficult because of the variety of paths taken by water transiting the system plus the fact that the >20 water control gates on the area are not individually gauged. More importantly, most water entering Eagle Bluffs is lost to infiltration and we lack data on the potentially crucial effect on nutrients of passage through the soil-water interface where phosphorus removal and denitrification are potentially most intense (Howard-Williams 1985). Monitoring in the McBaine Bottoms has included extensive groundwater studies by the U.S.G.S. (Richards 1995), but this work is focused on drinking water well fields north of Eagle Bluffs rather than on the main local flow paths of groundwater which are presumably downriver to the south and east (Fig. 1). We have recently discovered an extensive series of groundwater seeps on the banks of Perche Creek of the eastern boarder of Eagle Bluffs. In combination with U.S.G.S. data, information from these seeps may eventually permit more quantitative characterization of nutrient removal by the Eagle Bluffs wetlands.

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Table 4.–Nutrients and chlorophyll from Missouri River oxbow lakes and the Distribution Channel and terminal pools on the Eagle Bluffs Conservation Area. The oxbow data are from monthly, year-around samples collected in June 1994 – May 1998 (Knowlton and Jones 1997). Means are shown ±1 standard deviation.

		Oxbows	DC	Pools
dTP ($mg \cdot L^{-1}$)	n	239	244	290
	mean	0.08 ± 0.06	1.23 ± 0.62	0.53 ± 0.47
	range	0.01 - 0.31	0.24 - 2.87	0.03 - 2.22
dTN ($mg \cdot L^{-1}$)	n	239	244	290
	mean	0.86 ± 0.52	5.78 ± 4.43	2.37 ± 2.51
	range	0.21 - 4.08	0.59 - 21.55	0.36 - 13.01
CHL ($\mu g \cdot L^{-1}$)	n	287	242	285
	mean	129 ± 168	102 ± 158	146 ± 266
	range	1 - 1248	3 - 847	<1 - 3296

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