Fluoride and aquatic environmental impacts in Austin

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Abstract

Adverse impacts to aquatic life in Austin streams and rivers from use of fluoride in drinking water treatment are evaluated using relevant effect levels versus measured and predicted instream concentrations. At the present levels of fluoride addition, there does not appear to be a strong potential for adverse aquatic life impacts. However, the fluoridation of drinking water does contribute to the fluoride in Austin area streams. Due to the low potential for aquatic life impacts, the scientific evaluation of the need for continued fluoridation should be made on the basis of human health.

Introduction

Fluoride, though ubiquitous and naturally occurring in freshwater systems (ATSDR 1993), has been demonstrated to exert adverse impacts on aquatic life at elevated concentrations. This study compares the predicted and measured concentrations of fluoride in Austin's streams and rivers against relevant aquatic life effect levels. Mass balance modeling methods have been used to evaluate resulting instream concentrations of fluoride in local and regional scales (Osterman 1990). Health benefits and concerns from human consumption of fluoridated water must be considered separately by human health authorities as a separate set of criteria and testing literature would be applicable.

Fluoridated drinking water may enter the aquatic environment primarily via potable water landscape irrigation, leaking drinking water supply infrastructure and through treated wastewater effluent discharge (Osterman 1990). Significant amounts of fluoride can be removed in secondary wastewater treatment processes (Masuda 1964). Fluoride is not expected to concentrate in groundwater, and is likely to be filtered out as water infiltrates through soils (Pescod 1992). Fluoride is not readily leached from soils, and due to low solubility in soils is typically less available to terrestrial plants (WHO 2002).

Fluoride may be permanently bioaccumulative in some tissues like bone and scales (Neuhold and Sigler 1960), although large-scale impacts of biomagnification in aquatic food webs is unknown (ATSDR 1993). Fluoride toxicity varies inversely with calcium hardness (Smith et al 1985, WHO 2002) and temperature. Fluoride toxicity to aquatic organisms is highly species specific.

The Austin Water Utility maintains fluoride concentrations in finished drinking water with a control range of 0.6 to 0.8 mg/L. Fluoride additions at drinking water plants monitor existing fluoride inputs to insure fluoridation remains within acceptable control limits. For the purposes of this review, a conservative 0.8 mg/L will be used to represent finished drinking water fluoride concentrations that could be released to the aquatic environment through either leaking water supply infrastructure or irrigation in the aquatic environments upstream of Austin Water Utility wastewater effluent outfalls. Impacts of fluoride in Austin's wastewater effluent on the Colorado River are evaluated separately.

The Austin Water Utility is currently permitted by the Texas Commission on Environmental Quality to discharge approximately 150 mgd (232 ft^3/s) of treated wastewater to the Colorado River below Austin. The permitted flow rates are greater than actual flow rates, although permitted flow rates are considered for conservative estimates. Although fluoride concentrations are likely to be reduced by existing wastewater treatment processes, fluoride concentrations are expected to be higher in raw wastewater influents versus targeted drinking water concentrations. The Austin Water Utility does measure fluoride concentrations in treated effluent, and these measurements were included directly in this assessment. Ambient flows in the Colorado River downstream of Austin and upstream of any Austin municipal wastewater effluent outfall are represented by the United States Geological Survey gauge at the US183 bridge (USGS 08158000).

Environmental Effect Levels

Aquatic life lethal concentration for 50% of test organisms (LC_{50}) and other effect levels for fluoride in published literature vary greatly, and experimental conditions may not mimic regional differences in covariates like hardness and temperature. Some of the lowest available relevant effect levels are summarized in Table 1. Trout and salmon are likely to be more sensitive than other fish and are not present in our warm-water river systems, but are included as a lower reference point.

Table 1. Selected lowest available relevant effect levels for freshwater aquatic organisms from literature.

Organism	**Endpoint**	**Fluoride Conc. (mg/L)**	**Reference**
Rana pipiens (leopard frog)	30-day LOEC	5	Kaplan et al 1964
Rana pipiens (leopard frog)	development effects	1.0*	Kuusisto and Telkka 1961
Chlorella (algae)	growth effects	2.0	Groth 1975
Oncorhynchus mykiss (rainbow trout)	20-day LC_{50}	4.8	Angelovic et al 1961
Musculium transversum (fingernail clam)	8-wk LC_{50}	2.8	WHO 2002
Nitzschia palea (diatom)	some growth enhancement	> 30	Joy and Balakrishnan 1990
Hydropsyche bronta (caddisfly)	96-hr LC_{50}	17	Camargo et al 1992
Mysidopsis bahia (mysid shrimp)	96-hr LC_{50}	10.5	LeBlanc 1984
Pimephales promelas (fathead minnow)	96-hr LC_{50}	315	Smith et al 1985

*observed effects but no statistical verification

There are no State of Texas or U.S. Environmental Protection Agency (EPA) regulatory criteria for fluoride for the protection of aquatic organisms, although EPA human health drinking water maximum contaminant levels (MCL) are 4.0 mg/L with a secondary MCL of 2.0 mg/L. Lower recommended ambient fluoride levels (0.2 mg/L – 0.3 mg/L) for aquatic life have been established by the Government of British Columbia Ministry of Environment, although naturally occurring fluoride levels and water hardness are significantly lower in this area (EPD 1990). Aquatic organisms are likely to adapt to natural background fluoride concentrations, with brown trout found in the Madison River in Yellowstone National Park despite natural fluoride levels exceeding 12 mg/L (Neuhold and Sigler 1960).

Based on a review of available environmental effect levels for aquatic systems, it would appear that a lowest observed effect levels (LOEC) of 2.0 mg/L fluoride would be a reasonable screening value and that a no observed adverse effect concentration (NOEC) of 1.0 mg/L would be highly conservative. The Texas Surface Water Quality Standards (TSWQS) provide some information, but do not provide specific numerical criteria for many potential contaminants. For toxic materials where specific numerical criteria have not been established, TSWQS specify that the LC_{50} for the most sensitive aquatic organism may be used to derive a value protective of aquatic life as follows (30 TAC 307.6 (c)(7):

(A) acute criterion will be calculated as 0.3 of the LC50 concentration to the most sensitive aquatic species (LC50 x 0.3 = acute criterion);
(B) chronic criterion will be determined from appropriate chronic toxicity data or calculated as 0.1 of acute LC50 concentration to the most sensitive aquatic species (LC50 x 0.1 = chronic criterion.

Using the toxicity data in Table 1 as a guide, the lowest LC50 would be 2.8 mg/L making the estimated acute criteria 0.84 mg/L and the chronic criteria would be assumed as 0.28 mg/L. However, this LC50 is for a relatively long term testing period of 8 weeks. The more appropriate LC50 value to use as a basis for criteria may be the 96 hour LC50 for which the lowest value was 10.5 mg/L. Using this as the basis, the acute criteria would be 3.15 mg/L and chronic would be 1.05 mg/L. Since one of the measured observed effects level for developmental effects is 1.0 mg/L, using this for the NOEC and 2.0 for the LOEC would still appear to be a conservative estimate.

Monitoring Data

The City of Austin and the USGS have collected more than 3,455 measurements of fluoride in the Colorado River from Lake Travis downstream to Lady Bird Lake (upstream of Austin's wastewater effluent discharges) since 1967, primarily analyzed by SM 4500 (APHA 2005). Ambient levels of fluoride in aquatic systems vary greatly depending on regional-specific factors (ATSDR 1993). Analyses are mixed with fluoride measurements in both the dissolved and total fractions.

Based on available monitoring data, the average background fluoride concentrations in the Colorado River thru Austin is 0.21 mg/L (stdev = 0.04). The 99^{th} percentile of fluoride measurements in the Colorado River thru Austin is 0.31 mg/L, with a maximum recorded value (Lady Bird Lake at Longhorn Dam, 19 Jan 1988) of 0.7 mg/L. Flow patterns through the Colorado River change drastically based on downstream agricultural demands, with summer release periods yielding flows typically an order of magnitude higher than winter non-release periods. There is no statistically significant difference in mean fluoride concentrations in the Colorado River thru Austin between the high flow release (0.21 mg/L±0.03 ,n=2196) and low flow non-release period (0.21 mg/L±0.04, n=1259), suggesting that even with minimal dilution finished drinking water releases are not altering fluoride concentrations in source waters.

In addition to ambient fluoride measurements in the Colorado River, there are 1,334 fluoride measurements in surface waters from 1968 to present at 69 different locations throughout Austin. As in the Austin's drinking water sources, the median fluoride concentration is 0.2 mg/L with more than 97% of all measurements less than the 1.0 mg/L NOEC. There are 31 measurements with fluoride values above 1.0 mg/L, and all but 2 of these 31 measurements were historical values from the mouth of Walnut Creek downstream of the Walnut Creek wastewater treatment facility discharge (and prior to the relocation of that outfall directly to the Colorado River). Therefore, it can be assumed that these data are effluent dominated for a large part of the year. For larger watersheds that transition longitudinally from lesser developed headwaters to more urban lower reaches (e.g. Barton Creek, Bull Creek and Onion Creek), there are no differences in fluoride concentrations with increasing development. Lawn irrigation and drinking water supply line leaks or breaks are expected to increase with density and age of development. Background concentrations of fluoride in lesser developed creeks are consistent with mean values in the Colorado River thru Austin (0.2 mg/L).

Approximately 1,370 measurements of fluoride in 137 groundwater springs across Austin were assessed. Fluoride in springs is generally consistent with background levels measured in surface waters (mean=0.22 mg/L±0.15), with no measurements greater than 2.0 mg/L and 99% of all measurements less than 0.8

mg/L. Although ion concentrations in groundwater springs have been demonstrated to increase with development (Herrington et al 2007), this is not necessarily indicative of fluoridated drinking water impacts to groundwater. Fluoride concentrations in some rural springs (e.g., Holman Hollow, n=21, mean=0.2 mg/L ± 0.05) are not significantly different from fluoride in some springs in older urban areas (e.g., Stillhouse Hollow, n=29, mean=0.16 mg/L ± 0.1). However, fluoride concentrations may be slightly elevated in springs strongly influenced by irrigation in the contributing zones with treated wastewater effluent (e.g., Driving Range Spring, n=22, mean=0.31 mg/L ± 0.11). Fluoride concentrations may be increasing over time in Barton Springs (Hiers and Herrington 2008), although the maximum concentrations measured to date (n=257) is 0.65 mg/L and less than the NOEC.

Fluoride measurements in 125 different groundwater wells in Austin (n=834, mean=0.44 mg/L ± 0.62) in general yield higher concentrations than observed in groundwater springs, with some wells influenced by the Trinity Aquifer yielding fluoride values above the drinking water MCL of 4.0 mg/L. The majority of the measured values (96%), however, were below the secondary drinking water MCL of 2.0 mg/L. Geologic variations must be accounted for via multivariate analyses like Piper plotting and examination of the formation in which the groundwater is being evaluated to fully investigate differences in groundwater source contributions. A review of fluoride in groundwater previously conducted (Hauwert and Vickers 1994) identified the strong influences of interactions with minerals naturally occurring in Glen Rose limestone or leakage from the Trinity Aquifer on background fluoride levels in groundwater. Despite this variation, the general consistency of spring fluoride samples in Edwards limestone at concentrations less than the NOEC again suggest low potential for adverse environmental impacts. Measurements in cave drips, or surface waters that have infiltrated through the overlaying soil/rock matrix to underground conduits, appear to confirm the reduction of water column fluoride by interactions with soils as measured concentrations (n=49, mean=0.05 mg/L ±0.03) are less than surface background levels.

Colorado River flows estimated from USGS gauge 08158000 from 1992 to 2009 (to reflect current water management policies) yield median values of 1,050 ft^3/s and seven-day average low flows with 2-year return intervals (7Q2) of 146 ft^3/s. The full permitted volume of wastewater discharge from the City of Austin of 150 mgd (232 ft^3/s) is approximately equivalent to the 16^{th} percentile of flow. Thus, less than 16% of the period of record would yield a dilution factor of less than 1:1. These low flow periods typically occur during dry rainfall years in the winter months when demand from downstream agricultural uses is lower. There have been approximately 40 measurements of fluoride in Austin's wastewater effluent (prior to discharge) from the two major treatment plants which discharge directly to the Colorado River since 2001. The mean fluoride concentration in the treated effluent from the major plants was approximately 2.10 (±0.61) mg/L (non-detect values taken at detection limits for conservative estimate). With dilution at 7Q2 flow values, the resulting instream concentration is predicted by mass balance to be 1.37 mg/L, and at median flows the fluoride concentration is predicted to be 0.55 mg/L. More than 75% of the year fluoride concentrations from Austin's wastewater effluent are predicted to be diluted to 1.0 mg/L or less, and Colorado River flows would have to be less than 13 ft^3/s for the resulting instream concentrations to be 2.0 mg/L or higher after mixing. Daily average flow in the Colorado River has only been less than 13 ft^3/s on 18 days since 1900, and only one day since the current flow management policy was instituted in 1992 (representing less than 0.02% of the record). Based on average fluoride concentrations in the effluent and dilution, wastewater effluent impacts from fluoride on the Colorado River again do not appear to have a strong potential for negative aquatic life impacts.

As a an additional assessment of fluoride concentrations in the Colorado River downstream of Austin's wastewater discharges, the USGS historically measured fluoride upstream of Bastrop (USGS site 08159200), with 171 measurements of dissolved fluoride from 1968 to 1994. The average value was 0.35 mg/L (±0.15), although the time period is clearly not reflective of the most current conditions. The maximum value was 0.9 mg/L (10 Dec 1968), and 99% of all values were less than 0.83 mg/L. Although wastewater effluent may be slightly increasing the concentration of fluoride in the Colorado River, values are still expected to be below the LOEC of 2.0 mg/L for all nominal conditions.

In comparison to the 0.8 mg/L targeted fluoridation concentration of drinking water, ambient monitoring data particularly in surface waters yield lower background concentrations. Although the release of finished drinking water to aquatic environments could increase the concentration of fluoride in Austin streams (and would definitely increase the mass loading of fluoride), there is no clear indication that ambient fluoride concentrations are being noticeably increased. By mass balance there would be less than or equal to 0.8 mg/L fluoride as any existing ambient flows (which would have lower background concentrations of fluoride) would dilute the drinking water fluoride. Comparison of the maximum predicted 0.8 mg/L fluoride concentrations in receiving streams as well as ambient monitoring data to the effects levels reviewed and the suggested NOEC of 1.0 mg/L fluoride clearly demonstrates the low potential for adverse environmental impacts to aquatic systems in the Austin area.

Discussion

Fluoride increases in ambient water is not likely to have adverse impact in Austin streams at current levels. The finding of a lack of obvious potential for environmental degradation is consistent with other national reviews (Osterman 1990; Pollick 2004). However, it is likely that some mass loading increase of fluoride would occur as a result finished drinking water releases to aquatic environments. Minimization of anthropogenic sources of contamination to aquatic systems is environmentally preferable as in-situ cumulative and synergistic impacts may not always be fully understood. Additionally, the processing and transport of fluoride may have additional indirect environmental effects. Thus, from a purely environmental protection standpoint there is no reason to fluoridate drinking water. The decision to fluoridate drinking water should be based on the comparison of human health benefits from reduced dental caries versus any potential human health risks (Kumar 2008).

References

Agency for Toxic Substances and Disease Registry (ATSDR). 1993. Toxicological profile for fluorides, hydrogen fluoride, and fluorine. US Department of Health and Human Services (TP-91/17).

American Public Health Association (APHA). 2005. Standard Methods for the Examination of Water and Wastewater. 20th edition.

Angelovic, J. W., and W. F. Sigler, J. M. Neubold. 1961. Temperature and Fluorosis in Rainbow Trout. Journal of the Water Pollution Control Federation (33):371.

Camargo, J.A., and J.V. Ward, K.L. Martin. 1992. The relative sensitivity of competing hydropsychid species to fluoride toxicity in the Cache la Poudre River (Colorado). Archives of Environmental Contamination and Toxicology (22): 107–113.

Environmental Protection Division (EPD), Government of British Columbia. 1990. Ambient water criteria for fluoride. Ministry of Environment.

Groth, E. 1975. An Evaluation of the Potential for Ecological Damage by Chronic Low-Level Environmental Pollution by Fluoride. Fluoride 8(4): 224-240.

Hauwert, N. and S. S. Vickers. 1994. Barton Springs/Edwards Aquifer Hydrogeology and Groundwater Quality. TWDB Grant Contract No 93-483-346. Prepared for the Texas Water Development Board, Submitted by the Barton Springs/Edwards Aquifer Conservation District.

Herrington, C., and S. Hiers, S. Pope, A. Clamann. 2007. Differential changes in groundwater quality due to urbanization under varying environmental regulations in Austin, Texas. City of Austin Watershed Protection Department, Environmental Resource Management Division, Water Resource Evaluation Section. SR-07-0D. DRAFT.

Hiers, S., and C. Herrington. 2008. Temporal trend analysis of long-term monitoring data at karst springs. City of Austin Watershed Protection Department, Environmental Resource Management Division, Water Resource Evaluation Section. SR-08-06. DRAFT.

Joy, C. M., and K. P. Balakrishnan. 1990. Effect of fluoride on axenic cultures of diatoms. Water Air Soil Pollution (49): 241–249.

Kaplan, H. M., and N. Yee, S. S. Glaczenski. 1964. Toxicity of Fluoride for Frogs. Laboratory Animal Care (14): 185-198.

Kumar, J. V. 2008. Is water fluoridation still necessary? Advances in Dental Research (20): 8-12.

Kuusisto, A. N., and A. Telkka. 1961. The Effect of Sodium fluoride on the Metamorphosis of Tadpoles. Acta Odontologica Scandinavica 19 (1): 121-127.

LeBlanc, G. A. 1984. Interspecies relationships in acute toxicity of chemicals to aquatic organisms. Environmental Toxicology and Chemistry (3): 47–60.

Masuda, T. T. 1964. Persistence of fluorides from organic origins in waste waters. Developments in Industrial Microbiology (5): 53-70.

Neuhold, J. M., and W. F. Sigler. 1960. Effects of Sodium Fluoride on Carp and Rainbow Trout. Transactions of the American Fisheries Society (89): 358-370.

Osterman, J. W. 1990. Evaluating the impact of municipal water fluoridation on the aquatic environment. American Journal of Public Health (80): 1230-1235.

Pescod, M. B. 1992. Wastewater treatment and use in agriculture—FAO irrigation and drainage paper 47. Food and Agriculture Organization of the United Nations. Wastewater use case studies T0551E.

Pollic, H. F. 2004. Water Fluoridation and the Environment: Current Perspective in the United States. International Journal of Occupational and Environmental Health (10): 343-350.

Smith, L. R., and T. M. Holsen, N. C. Ibay, R. M. Block, A. B. DeLeon. 1985. Studies on the Acute Toxicity of Fluoride Ion to Stickleback, Fathead Minnow, and Rainbow Trout. Chemosphere 14(9): 1383-1389.

World Health Organization (WHO). 2002.   Fluorides. Environmental Health Criteria 227. United Nations Environment Programme.