Impacts of Bromide from Power Plants on Downstream Disinfection Byproduct Formation

A Literature Review

3002017477

Impacts of Bromide from Power Plants on Downstream Disinfection Byproduct Formation

A Literature Review

3002017477

Technical Update, November 2019

EPRI Project Managers

N. Goodman

J. Thomas

ELECTRIC POWER RESEARCH INSTITUTE
3420 Hillview Avenue, Palo Alto, California 94304-1338 ▪ PO Box 10412, Palo Alto, California 94303-0813 ▪ USA
800.313.3774 ▪ 650.855.2121 ▪ askepri@epri.com ▪ www.epri.com

THE ELECTRIC POWER RESEARCH INSTITUTE (EPRI) PREPARED THIS REPORT.

This is an EPRI Technical Update report. A Technical Update report is intended as an informal report of continuing research, a meeting, or a topical study. It is not a final EPRI technical report.

NOTE

For further information about EPRI, call the EPRI Customer Assistance Center at 800.313.3774 or e-mail askepri@epri.com.

ACKNOWLEDGMENTS

The Electric Power Research Institute (EPRI) prepared this report.

Principal Investigators
N. Goodman
J. Thomas

This report describes research sponsored by EPRI.

EPRI acknowledges the assistance of the following individuals in preparing this report:

Coco Liu – EPRI

Katherine Dombrowski –Trimeric Corporation

Kathy Hall, Laura Weintraub, Renn Lambert – LimnoTech

This publication is a corporate document that should be cited in the literature in the following manner:

Impacts of Bromide from Power Plants on Downstream Disinfection Byproduct Formation: A Literature Review. EPRI, Palo Alto, CA: 2019. 3002017477.

ABSTRACT

Several recent journal articles developed mass balance models of bromide use in coal-fired power plants and modeled impacts of bromide in flue gas desulfurization (FGD) wastewater on surface water quality and formation of disinfection byproducts in drinking water treatment systems. EPRI conducted a detailed review of several of these articles, focusing on the methodology used to estimate the downstream bromide concentrations and potential health risks. The assumptions used in the articles were evaluated based on information obtained from recent surveys of power plant bromide usage and from a more detailed evaluation of the bromide content of coals commonly burned for power production in the United States.

Keywords

Bromide

Disinfection byproducts

Drinking water

Flue gas desulfurization

Trihalomethanes

ELECTRIC POWER RESEARCH INSTITUTE

EXECUTIVE SUMMARY

Deliverable Number: 3002017477

Product Type: Technical Update

Product Title: Impacts of Bromide from Power Plants on Downstream Disinfection Byproduct Formation: A Literature Review

---

PRIMARY AUDIENCE: Environmental managers of coal-fired power plants

SECONDARY AUDIENCE: Environmental regulators and the public

KEY RESEARCH QUESTION

The objective of this project is to evaluate for accuracy the methodologies used in several recent peer-reviewed articles to model the impact of bromide releases from coal-fired power plants on the formation of disinfection byproducts in downstream drinking water treatment systems (DWTSs).

RESEARCH OVERVIEW

EPRI reviewed several articles that modeled the discharge of bromide from power plant flue gas desulfurization (FGD) wastewater discharges, transport to downstream drinking water treatment intakes, and potential human health risks due to formation of disinfection byproducts. Modeling the source term and transport of substances for even a single well-characterized source and surface water body is difficult; modeling at the watershed or national level requires many simplifying assumptions. The use of conservative assumptions to compensate for uncertainty is a common approach in health risk assessments, where the intent is to be protective of human health, but can lead to unrealistic outcomes when multiple conservative assumptions are combined. EPRI's review is intended to improve the accuracy of such Br discharge modeling and inform future modeling efforts.

The following articles were the primary focus of this project:

Cornwell, D. A., Baljit, K. S., Brown, R., and McTigue, N. E. 2018. "Modeling Bromide River Transport and Bromide Impacts on Disinfection Byproducts." *Journal of the American Water Works Association*, Volume 110, Issue 11, November.

Good, K. D. and VanBriesen, J. M. 2016. "Current and Potential Future Bromide Loads from Coal-Fired Power Plants in the Allegheny River Basin and Their Effects on Downstream Concentrations." *Environ. Sci. Technol.* 2016, 50, 9078−9088.

Good, K. D. and VanBriesen, J. M. 2017. "Power Plant Bromide Discharges and Downstream Drinking Water Systems in Pennsylvania." *Environ. Sci. Technol*. 2017, 51, 11829−11838.

Together...Shaping the Future of Electricity®

Electric Power Research Institute
3420 Hillview Avenue, Palo Alto, California 94304-1338 • PO Box 10412, Palo Alto, California 94303-0813 USA
800.313.3774 • 650.855.2121 • askepri@epri.com • www.epri.com

Good, K. D. and VanBriesen, J. M. 2019. "Coal-Fired Power Plant Wet Flue Gas Desulfurization Bromide Discharges to U.S. Watersheds and Their Contributions to Drinking Water Sources." *Environ. Sci. Technol.*, 53, 213–223 and Supporting Information.

Regli, S. et al. 2015. "Estimating Potential Increased Bladder Cancer Risk Due to Increased Bromide Concentrations in Sources of Disinfected Drinking Waters." *Environ. Sci. Technol.*, 49, 13094−13102 and Supporting Information.

Additional literature and data sources were consulted as needed to evaluate the methodology and conclusions of these articles.

KEY FINDINGS

The authors used information available at the time of publication to estimate downstream impacts for a large number of U.S. coal-fired power plants. Because of more recent retirements of coal-fired units, changes in power industry practices, and the use of conservative simplifying assumptions in the models, the articles overestimate the regional or national impact of power plant bromide discharges. In particular:

- One-quarter of the power plants modeled by Good and VanBriesen (2019) and three of seven plants on the Ohio River modeled by Cornwell et al. (2018) are no longer in operation or do not discharge FGD wastewater to surface water.
- Based on recent surveys of power companies, the amount of bromide added to refined coal or for mercury control in recent years by power plants that discharge to surface water is lower than assumed by Good and VanBriesen (2019) and far lower than assumed by Cornwell at al. (2018).
- On a nationwide basis, the concentrations of bromide in coal are lower than the values used by Good and VanBriesen (2019) to model downstream population vulnerability and much lower than assumed by Cornwell et al. (2018).
- Good and VanBriesen (2016, 2019, and 2017 by reference) present an equation with an error in calculating coal tonnage, which—if actually used in modeling bromide mass loadings to surface water—would overestimate the mass of bromide by 14% for bituminous coal, 88% for subbituminous coal, 130% for lignite coal, and 42% for refined coal. Cornwell et al. (2018) cites the Good and VanBriesen (2016) methodology; therefore, it is possible that Cornwell et al. (2018) also contains the error. EPRI has received confirmation that the Good and VanBriesen (2016) article has this error and will continue to investigate whether it impacts the conclusions of the other articles and the model results.
- Good and VanBriesen (2019) procedures for estimating bromide concentrations in native and refined coal based on coal rank overestimate mass loadings to surface water by 5–58% for the modeling scenario used to evaluate downstream population vulnerability. These overestimates are in addition to the overestimates associated with the error in the coal tonnage calculation.
- The hydrologic modeling performed by Good and VanBriesen (2019) used a modeling approach that may have underestimated dilution. Cornwell et al. (2018) focused on low flow conditions, which may be appropriate for modeling potential short-term exceedances of drinking water disinfection byproduct (DBP) limits but not for evaluating long-term health risk. Neither model was validated with in-stream monitoring data.
- The assumption by Good and VanBriesen (2019) that the bromide concentration in water at the intake of a drinking water treatment plant is proportional to population risk is not valid. Many factors within the treatment plant and in the water distribution system influence the levels of DBPs reaching the consumer.
- The surface water bromide concentrations that Regli et al. (2015) and Good and VanBriesen (2019) consider significant for cancer effects from exposure to trihalomethanes (THMs) and population vulnerability to DBP in drinking water are within the range of concentrations observed in relatively uncontaminated (reference) water bodies. The fact that it is very common for surface waters to exceed the calculated thresholds of concern suggests that additional research is needed on sources of bromide and the relationship among bromide, DBP formation, and health effects.

- The use of conservative assumptions to compensate for uncertainty in complex watershed scale modeling of fate and transport of pollutants is a common approach in health risk assessments, where the intent is to be protective of human health. This report stresses caution with this approach because it can lead to unrealistic outcomes when multiple conservative assumptions are combined. It is hoped that the observations in this report will assist in improving future modeling efforts by identifying areas in which more recent data (for example, bromide addition rates), a more detailed analysis of existing data (such as coal subrank shipments), a more sophisticated approach (hydrologic modeling), and validation of the models would improve the analysis.

A more accurate approach than modeling to determine impacts of upstream bromide sources on a downstream drinking water facility is a watershed approach, using monitoring data from each source, site-specific hydrologic data, and measurement of bromide and DBPs in downstream drinking water treatment system influent and water delivery systems.

WHY THIS MATTERS

The element bromine and its ionic form bromide are common in the environment. Sources of bromide include natural geologic formations (seawater and saline groundwater), industrial sources such as coal combustion and oil recovery wastewaters, and municipal sources such as road salt and municipal wastewater. When water is chlorinated during drinking water disinfection, bromide in the water can attach to natural organic matter, forming brominated DBPs. Bromide is present in coal, and some power plants add bromide to the coal or boiler to improve the capture of mercury in pollution control devices. When flue gases pass through a wet FGD scrubber designed to capture gases such as sulfur dioxide, a portion of the bromide is captured in the scrubber water and may then be discharged to surface water. Owners of power plants, DWTSs, and the public would benefit from more accurate information on bromide concentrations, impacts of DWTS processes on DBP formation and removal, and potential human health impacts.

HOW TO APPLY RESULTS

Conclusions of this report will assist power plant owners, decision makers from regulatory agencies, and the public in better understanding the potential impacts of bromide on DBP formation in drinking water treatment.

EPRI CONTACTS: Naomi Goodman, Senior Technical Leader, ngoodman@epri.com; Jeff Thomas, Senior Technical Leader, jthomas@epri.com

PROGRAMS: Water Quality Assessment, P196A; Effluent Guidelines and Wastewater Monitoring, P196B

ACRONYMS AND ABBREVIATIONS

BOP	balance of plant
DBP	disinfection byproduct
DWTS	drinking water treatment system
EEMS	emission-economic modeling system
EGU	electricity generating unit
EIA	Energy Information Agency
EPA	U.S. Environmental Protection Agency
EPRI	Electric Power Research Institute
ESP	electrostatic precipitator
FGD	flue gas desulfurization
GW	gigawatt
HAA	haloacetic acid
HAA4	haloacetic acids, 4 unregulated species
HAA5	haloacetic acids, 5 regulated species
HUC	hydrologic unit code
IRIS	Integrated Risk Information System
LRAA	locational running annual average
MATS	Mercury and Air Toxics Standards Rule
MCL	maximum contaminant level
MCLG	maximum contaminant level goal
PAC	powdered activated carbon
PPM	parts per million
SDWA	U.S. Safe Drinking Water Act
SWAT	surface water analytical tool
THM	trihalomethane
TTHM	total trihalomethanes
UCMR 4	4^{th} Unregulated Contaminant Monitoring Rule
USGS	U.S. Geological Survey
ZLD	zero liquid discharge

CONTENTS

LIST OF FIGURES

LIST OF TABLES

1
INTRODUCTION

Recent peer-reviewed articles have modeled the impacts of bromide contained in wastewater discharges from coal-fired power plants on the formation of disinfection byproducts (DBPs) in downstream drinking water treatment systems (DWTS). The objective of this literature review is to evaluate the modeling methodologies used to evaluate downstream impacts of bromide. Where specific power plants and watersheds are modeled, the review focuses on the technical approach rather than the conclusions related to a specific downstream drinking water facility.

Background

Bromide is not in itself toxic, but during disinfection of water with chlorine or chloramines and in the presence of natural organic matter, bromide can contribute to the formation of toxic halogenated DBPs such as trihalomethanes (THMs) and haloacetic acids (HAAs). These two classes of DBPs are regulated in drinking water by the U.S. Environmental Protection Agency (EPA) as shown in Table 1-1.

Table 1-1
U.S. drinking water standards for THMs and HAAs

Contaminant	MCL (μg/L)	Health Effect Basis
Total Trihalomethanes (TTHM) Bromodichloromethane Bromoform Dibromochloromethane Chloroform	80 (LRAA)	Liver, kidney or central nervous system problems; increased risk of bladder cancer
Five Haloacetic acids (HAA5) Dichloroacetic acid Trichloroacetic acid Monochloroacetic acid Bromoacetic acid Dibromoacetic acid	60 (LRAA)	Increased risk of cancer

MCL: Maximum Contaminant Level.
LRAA - locational running annual average

There is not a one-to-one relationship between the concentration of bromide in the influent to a DWTS and the concentration of DBPs in finished drinking water. The amount of DPBs formed depends on multiple factors in addition to the bromide concentration, such as the natural organic matter concentration and characteristics, the pH and temperature of the water, and the disinfectant dose and chemical used. The final drinking water DBP concentration also depends on the residence time of the water in the distribution system after secondary disinfection and the extent of removal of DBPs by loss to air (for THMs) and biodegradation (for HAAs). There is extensive literature on factors influencing DBP formation, which EPRI reviewed in an earlier report (EPRI, 2014a).

Sources of bromide from coal-fired power plants include bromide in the coal that is captured in wet flue gas desulfurization (FGD) systems designed to reduce air pollution from sulfur dioxide and acid gases. In some FGD scrubbers, liquid must be periodically purged to prevent corrosion of the equipment from built-up salts. However, not all plants with wet FGD systems discharge wastewater to surface water. Some plants can manage the FGD water balance such that the residual moisture remains in the FGD solids and does not need to be purged. Other plants may use alternative water handling methods such as evaporation, crystallization or deep-well injection. Plants that employ dry FGD systems produce no wastewater.

Bromide can be added to coal, to the boiler, or post-combustion to aid in the control of mercury emissions. In addition, some facilities use brominated anti-fouling agents to control biota growth in cooling water systems. The papers reviewed by EPRI consider only the coal and additives; that is the focus of EPRI's review as well. A complete evaluation of downstream impacts should consider all sources of bromide; however, EPRI (2014a) found no published information on the amounts or frequency of bromide biocide usage by power companies.

The EPRI literature review for this study concentrated on recently-published articles that modeled impacts of bromide discharged from coal-fired power plants, specifically from discharge of flue gas desulfurization (FGD) wastewater to surface water upstream of DWTS. The articles that were the primary focus of this project are:

> Good, K. D., and VanBriesen, J. M. Coal-Fired Power Plant Wet Flue Gas Desulfurization Bromide Discharges to U.S. Watersheds and Their Contributions to Drinking Water Sources, *Environ. Sci. Technol*., 53, 213-223, 2019.
>
> Regli, S. et al. Estimating Potential Increased Bladder Cancer Risk Due to Increased Bromide Concentrations in Sources of Disinfected Drinking Waters. *Environ. Sci. Technol*. 2015, 49, 13094−13102.
>
> Cornwell, D. A., Baljit, K. S., Brown, R., and McTigue, N. E. Modeling Bromide River Transport and Bromide Impacts on Disinfection Byproducts, *Journal of the American Water Works Association*, Volume 110, Issue 11, November 2018.

Earlier articles by Good and VanBriesen (2016; 2017) that estimated downstream impacts of bromide sources in the Allegheny River Basin (2016) and in Pennsylvania watersheds (2017) were also reviewed and are discussed below as necessary. Additional literature sources were consulted as needed to evaluate the methodology and conclusions of these articles.

Summary of Reviewed Articles

Good and VanBriesen (2019) identified coal-fired power plants upstream of drinking water treatment facilities that potentially discharged FGD wastewater to surface water. Bromide mass loadings to surface water were calculated for each Electricity Generating Unit (EGU) based on coal bromide content calculated from data from the U.S. Geological Survey (USGS)

COALQUAL database and from 2016 Energy Information Agency (EIA) records of coal ranks burned and refined coal usage. Power plants that added bromide or burned refined coal were identified from power company surveys conducted by EPRI prior to 2013 and from information provided by power companies to the EPA. An assumed bromide addition rate of 100 parts per million (ppm) was used to model bromide mass loadings from those EGUs. Good and VanBriesen (2019) assumed that 84% of bromide in the coal was captured in the FGD wastewater. Flow path modeling using median flow rates was used to estimate concentrations of bromide reaching downstream drinking water facilities from upstream power plants. No measurements of bromide concentrations in power plant discharges, water bodies, or DWTS inlets were used to develop or validate the model.

The criterion used to determine how far downstream to model bromide transport was developed from the conclusions of a paper by Regli et al. (2015) that related increases in bromide to increases in total trihalomethanes (TTHM) during water chlorination, and related increases in TTHM to increases in bladder cancer. A bromide concentration of 10 micrograms per liter (µg/L) was identified by Regli et al. (2015) as associated with an excess lifetime risk of bladder cancer of one in ten thousand (10^{-4}) attributed to TTHM exposure. Good and VanBriesen (2019) divided the Regli et al. value by 10 to account for use of median river flow rates; flow modeling was continued downstream until the increase in bromide at the intake of any DWTS fell below 1 µg/L.

Good and VanBriesen (2019) identified watersheds with "significant downstream effects" by summing the modeled downstream bromide concentration contributions at each DWTS from all upstream power plants. That bromide amount was then multiplied by the population served by the DWTS to calculate a "population effect" value.

Regli et al. (2015) related an increase in surface water bromide to an increase in formation of total trihalomethanes (TTHM) during chlorination, and then derived an association between the change in TTHM (ΔTTHM) with an increase in lifetime risk of bladder cancer. The risk assessment had two components:

- Changes in TTHM with increases in bromide were estimated using an empirical model of disinfection processes and DBP concentrations developed by the EPA for the Stage 2 Disinfectants and Disinfection Byproducts Rule.
- The additional risk of bladder cancer associated with a given increase in TTHM was calculated from epidemiological studies.

The results of these two evaluation steps were combined to reach the conclusion that an increase of 10 µg/L bromide in the inlet to a DWTS was the lowest level at which a significant increase in TTHM would be observed. A significant increase was concluded to be any increase over 1 µg/L TTHM, which Regli et al. (2015) calculated to produce an increased lifetime risk of bladder cancer of 10^{-4}.

Cornwell et al. (2018) modeled formation of TTHM and haloacetic acids (HAAs) of bromide discharged by power plants in two watersheds, seven plants on the Ohio River (IN and KY) and one plant on the Dan River (NC). EPRI's focus in reviewing this paper was on the modeling methodology for the Ohio River, as actual discharge concentrations and river hydrologic data were used in the Dan River evaluation. Evaluation of site-specific data on power plant discharges, river hydrology, and downstream drinking water facility practices was outside the scope of this literature review.

Cornwell et al. (2018) modeled a “worst-case” scenario consisting of low flow, high bromide native coal and a high bromide addition rate. The model used 2014 hydrologic data to determine a worst-case potential impact on DBP formation. To estimate native coal concentrations for the Ohio River plants, the paper used bituminous coal chloride data from the USGS COALQUAL database and a Br to Cl ratio of 0.02 from Kolker and Quick (2015). The bituminous coal fired at these power plants was assumed to have 300 mg/kg of bromide added for mercury control. The paper states that the methodology of Good and VanBriesen (2016) was used to calculate bromide mass loadings to surface water. Other than a few very general comparisons to existing data, no measurements of bromide concentrations in power plant discharges, water bodies, or DWTS inlets were used to develop or validate the model for the Ohio River.

After reviewing numerous published models relating bromide concentration to formation of DBPs during chlorination, Cornwell et al. (2018) developed models from laboratory jar studies in which different levels of bromide were spiked into samples from the source water of 13 drinking water treatment plants and the samples were treated with chlorine or chloramines. The jar studies attempted to replicate disinfection doses and exposure times from the source DWTS, but the article did not state whether commonly applied measures to reduce DBP levels in the finished water such as enhanced coagulation or powdered activated carbon removal were included in the experimental design. Regression equations were developed relating increases in bromide to increases in TTHM, HAA5 (five regulated HAAs) and HAA4 (four unregulated, brominated HAAs). Cornwell et al. (2018) suggested that the regression equations could be useful on a nationwide basis to predict potential increases in DBPs. However, the regression models fit to the data (r^2 ranging from 0.22 to 0.65) indicate that there were factors not considered in the regression terms that influence DBP formation. In addition, this approach did not consider processes within the DWTS and water distribution system that can reduce DBPs before they reach the consumer. The paper concluded that under low-flow conditions on the Ohio River, the increase in bromide concentration at DWTS within 50 miles downstream of a power plant could exceed 50 µg/L for 170 days per year.

<u>Good and VanBriesen (2016 and 2017)</u> used COALQUAL chloride data and a chloride/bromide ratio of 0.02 to calculate native coal bromide mass loadings exiting power plants power plants with wet FGDs and model downstream bromide concentrations at drinking water intakes. A Monte Carlo analysis rather than a single set of values was used to estimate low, medium, and high estimates of monthly bromide loadings to surface water. The 2016 paper modeled four power plants in the Allegheny River basin. The 2017 paper modeled nine power plants in Pennsylvania and West Virginia that were upstream of drinking water intakes in Pennsylvania.

Report Organization

Because similar approaches were used in the reviewed articles to model downstream impacts, this report has been organized by topic rather than focusing on the individual articles. The topics discussed in the following sections are as follows:

Section 2 – Bromide in Native Coal – discusses the approaches used to estimate bromide content in the coal burned in power plants.

Section 3 – Bromide Addition – evaluates assumptions as to the amount of bromide added to coal or to the power plant boiler to promote mercury removal and/or to meet the refined coal specifications of Section 45 of the U.S. Internal Revenue Service tax code.

Section 4 – Partitioning – examines the models' assumptions on the percentage of bromide in coal that ends up in the FGD wastewater.

Section 5 – Number of FGD Wastewater Discharges – reviews the status of operating coal-fired units with wet FGD systems as of December 31, 2018 and compares those to the number of units assumed by Good and VanBriesen (2016; 2017; 2019) and Cornwell et al. (2018).

Section 6 – Bromide Loads to Surface Water – reviews the methods and assumptions used to estimate mass flows in pounds per year from power plants to surface water.

Section 7 – Hydrologic Modeling – evaluates the downstream transport models used in each of the reviewed articles.

Section 8 – Health Risk Assessment – reviews the methodology and conclusions of the toxicity and risk evaluation from Regli et al. (2015).

Section 9 – Bromide Concentrations of Concern – compares the conclusions of the reviewed papers regarding the modelled levels of bromide that are problematic for downstream impacts with bromide monitoring data from publicly available sources.

Section 10 – Potentially Affected Populations – reviews the approach taken by Good and VanBriesen (2019) to develop estimates of populations potentially impacted by DBPs associated with power plant bromide discharges.

Section 11 – Conclusions – discusses the overall findings of the literature review.

Section 12 – References – provides citations to sources of information cited in this report.

2
BROMIDE IN NATIVE COAL

Both Good and VanBriesen (2019) and Cornwell et al. (2018) estimated bromide inputs to power plants using coal data from the U.S. Geological Survey (USGS) COALQUAL database. Good and VanBriesen (2019) used bromide data directly to estimate power plant discharges for several coal types. Good and VanBriesen (2016; 2017) and Cornwell et al. (2018) applied a fixed ratio of bromide to chloride to COALQUAL chloride concentrations to estimate bromide.

The COALQUAL database is the largest public repository of bromide data for U.S. coals. The samples were obtained from coal beds and may not represent concentrations of coals received at the power plants; however, there is no comparable source of as-received coal bromide data. For chloride, there are large collections of as-received coal analyses that were produced in response to the Information Collection Request (ICR) for EPA regulatory actions in 1999 and 2009. For bituminous coal, these data sources are more representative of as-received coals than COALQUAL.

Good and VanBriesen (2019)

Good and VanBriesen (2019) calculated coal bromide content of power plant fuels using the *apparent rank* field in COALQUAL. These apparent ranks (e.g., HighVolatile bituminous A) will be referred to here as "subranks"[1]. The COALQUAL apparent rank field includes five subranks for bituminous coal, three for subbituminous coal, and two for lignite. Sample records for all subranks within a rank were then aggregated to calculate 25th, 50th, and 75th percentiles of bromide content for each coal rank (bituminous, subbituminous, and lignite coal).

The drawback of this approach is that the number of samples of each coal subrank analyzed for bromide in the COALQUAL database does not correlate to the usage of that coal subrank in coal-fired power plants. Bromide concentrations vary by subrank, especially for bituminous coal. For example, the low-volatile and medium-volatile bituminous subranks together account for approximately 15 percent of the bituminous coal bromide analyses in COALQUAL, but represent less than 2 percent of the tonnage of bituminous coal shipped to U.S. Electricity Generating Units (EGUs) in 2017 (Quick, 2019). High volatile bituminous A samples account for 61 percent of COALQUAL bromide analyses but just 35 percent of the tonnage of bituminous coal shipped to EGUs in 2017. Thus, the methodology used by Good and VanBriesen (2019) biased the coal bromide statistics by over- or under-representing subranks in the calculations.

---

[1] The rank of coals (bituminous, subbituminous, lignite) is defined by ASTM based on fixed carbon content, volatile matter content, and heating value. Bituminous coal rank is subdivided into high volatile C, B, and A; medium-volatile; and low-volatile groups on the basis of increasing heat value and fixed carbon content and decreasing volatile matter. Likewise, subbituminous rank is divided into groups C, B, and A on the basis of increasing heat value, and lignite is divided into groups B and A on the basis of increasing heat value (Wood et al., 1992).

To quantify the approximate degree of potential bias, EPRI calculated the 25th percentile, median, and 75th percentile bromide concentrations for each coal rank by weighting the subrank bromide concentration statistics by the tonnage of each subrank shipped to EGUs in 2017. The subrank shipment tonnages were provided to EPRI by Quick (2019), calculated from publicly available 2017 shipment data and heating values from the EIA Form 923 "Fuel Receipts and Costs" page. Tables 2-1, 2-2, and 2-3 compare the Good and VanBriesen (2019) bromide concentrations with the subrank-weighted values calculated by EPRI for bituminous, subbituminous, and lignite ranks, respectively[2]. The greatest bias was for bituminous coal.

- For bituminous coal, the differences in bromide content calculated by Good and VanBriesen (2019) and EPRI were substantial. While the 25th percentile values were within 0.5 parts per million (ppm), the Good and VanBriesen median values were 4.4 ppm higher than the values calculated by EPRI (45% higher) and the 75th percentile values were 10 ppm higher (58% higher).
- For subbituminous coal, the bromide concentrations determined by Good and VanBriesen (2019) and EPRI were similar, within 0.5 ppm. At the 75th percentile, the Good and VanBriesen bromide was 5.5 percent greater than the value calculated by EPRI.
- For lignite coal, the difference in bromide concentration statistics between the subranks was slightly more than for subbituminous coal. The lignite rank statistics determined by Good and VanBriesen (2019) were within 0.1 ppm for the 25th percentile, within 1 ppm for the median, and within 2 ppm for the 75th percentile. However, on a percentage basis, the Good and VanBriesen 75th percentile for lignite coal was 50% greater than the value calculated by EPRI.

Good and VanBriesen (2016 and 2017)

The Good and VanBriesen papers from 2016 and 2017 estimated native bromide concentration in bituminous coal by applying a ratio of 0.02 to COALQUAL chloride data. Values of 400, 550, and 2,900 ppm were used as the minimum, most likely, and maximum coal chloride concentrations. The resulting bromide concentrations used as input to the Monte Carlo simulation were 8, 11, and 58 ppm, respectively. This approach resulted in P25 and P75 values for bituminous coal that were approximately twice the estimates developed in the more detailed analysis conducted by Good and VanBriesen (2019).

---

[2] Throughout this report, halogen concentrations in coal are expressed on a dry basis.

Table 2-1
Bituminous coal bromide content, based on coal rank and on subrank-weighted bromide data

ASTM Class and Group (subrank)	Number of COALQUAL Br Analyses[1]	% of COALQUAL Br Analyses[1]	Short Tons Shipped to EGUs in 2017[2]	% of BIT tons shipped	P25 Br, ppm[1]	Median Br, ppm[1]	P75 Br, ppm[1]
Low volatile BIT	208	5.1%	1,358,832	0.6%	24.20	39.10	50.90
Medium volatile BIT	398	9.8%	2,945,196	1.3%	22.00	39.10	54.00
High volatile A BIT	2,476	61%	77,634,674	35%	4.04	13.90	25.30
High volatile B BIT	715	18%	52,436,943	24%	4.46	9.06	17.50
High volatile C BIT	268	6.6%	86,455,266	39%	2.08	4.17	8.33
Total BIT	4,065	100%	220,830,911	100%			
Good and VanBriesen (2019) bromide (subrank samples aggregated for each rank)					4.20	13.8	27.4
Bromide content weighted by coal subrank shipments to power plants					3.74	9.4	17.3
Good and VanBriesen (2019) bias compared to subrank shipment-weighted bromide					+12.3%	+45.3%	+58.4%

[1]Number of COALQUAL bromide analyses and bromide concentration statistics from Good and VanBriesen (2019).
[2] Subrank shipment classification provided by Quick, 2019.

Table 2-2
Subbituminous coal bromide content, based on coal rank and on subrank-weighted bromide data

ASTM Class and Group (subrank)	Number of COALQUAL Br Analyses[1]	% of COALQUAL Br Analyses[1]	Short Tons Shipped to EGUs in 2017[2]	% of SUBBIT tons shipped	P25 Br ppm[1]	Median Br, ppm[1]	P75 Br, ppm[1]
Subbituminous A	31	23.8%	16,856,231	5%	0.46	0.662	5.36
Subbituminous B	44	33.8%	46,457,356	13%	0.94	2.37	5.85
Subbituminous C	55	42.3%	286,691,138	82%	1.22	2.86	4.99
Total Subbituminous	130		350,004,725				
Good and VanBriesen (2019) bromide (subrank samples aggregated for each rank)					0.877	2.27	5.4
Bromide content weighted by coal subrank shipments to power plants					1.15	2.69	5.12
Good and VanBriesen (2019) bias compared to subrank-shipment weighted bromide					-23.7%	-15.6%	+5.5%

[1]Number of COALQUAL bromide analyses and statistics from Good and VanBriesen (2019).
[2] Subrank shipment classification provided by Quick, 2019.

Table 2-3
Lignite coal bromide content, based on coal rank and on subrank-weighted bromide data

ASTM Class and Group (subrank)	Number of COALQUAL Br Analyses[1]	% of COALQUAL Br Analyses[1]	Short Tons Shipped to EGUs in 2017[2]	% of LIG tons shipped	P25 Br, ppm[1]	Median Br, ppm[1]	P75 Br, ppm[1]
Lignite A	28	60.9%	56,273,380	95.9%	1.78	2.27	3.49
Lignite B	18	30.1%	2,416,657	4.1%	2.56	4.36	7.87
Total lignite	46	100%	58,690,037	100%			
Good and VanBriesen (2019) bromide (subrank samples aggregated for each rank)					1.93	2.99	5.5
Bromide content weighted by coal subrank shipments to power plants					1.81	2.36	3.67
Good and VanBriesen (2019) bias compared to subrank-shipment weighted bromide					+6.6%	+26.7%	+49.9%

[1]Number of COALQUAL bromide analyses and statistics from Good and VanBriesen (2019).
[2] Subrank shipment classification provided by Quick, 2019.

Cornwell et al. (2018)

Cornwell et al. (2018) adopted the same approach used in Good and VanBriesen (2016; 2017) of using COALQUAL chloride data to estimate bromide mass loadings from power plant FGD wastewater discharge. Seven power plants on the Ohio River were modeled, all of which fired bituminous coal. Although the paper also modeled bromide transport in the Dan River in North Carolina, that analysis used actual plant discharge data rather than assumed bromide loadings. Cornwell et al. (2018) cited a range of chloride concentrations for bituminous coal of 400 to 2,900 ppm; no citation was provided for these chloride values. The authors then applied a Cl:Br ratio of 0.02 (Kolker and Quick, 2015) to the 2,900-ppm chloride value to estimate a worst-case coal bromide concentration of 58 ppm Br. Coal concentrations were converted from dry measurements using an average moisture content of 6.5%.

As shown in Figure 2-1, the maximum chloride concentration of 2,900 ppm is an outlier compared to COALQUAL chlorine and bromine data for coal produced in 110 US counties. For comparison, the 75th percentile value of bituminous coal bromide concentration modeled by Good and VanBriesen (2019), 27.4 ppm, is also shown in this plot. A 58-ppm concentration for bituminous coal is approximately twice the Good and VanBriesen (2019) 75th percentile value and even exceeds the 90th percentile concentration of 45.8 ppm. It is more than three times the 75th percentile value developed by EPRI from subrank shipments (17.3 ppm). Use of the 75th percentile value is conservative, and will overestimate discharge loadings for most facilities.

Moisture Content of Native Coal

Coal moisture was used to convert bromide concentrations from a dry to a wet basis. Good and VanBriesen (2019) assumed moisture contents of 6.5%, 27%, and 34% for bituminous, subbituminous, and lignite coal, respectively. Cornwell et al. (2018) assumed 6.5% for bituminous coal. These values are appropriate for the coal ranks.

Impact of Coal Blending

As seen in Figure 2-1, there can be large regional differences in coal bromide and chloride concentrations within a coal rank. Both Good and VanBriesen (2019) and Cornwell et al. (2018) relied on statistics derived from coal rank data that do not take these variations into account.

The seven Ohio River plants evaluated in Cornwell et al. (2018) fired coals coming from many states, including Illinois, Indiana, Kentucky, Ohio, Pennsylvania, Virginia, and West Virginia. Coals from Indiana and Illinois comprised 25% of the 2017 coal fired, while coals from Kentucky comprised 4% of the coal fired. As seen in Figure 2-1, there can be large regional differences in coal bromide and chloride concentrations within a coal rank. For example, deep basin Illinois coal is significantly lower in bromide than Eastern Kentucky Southwestern Reserve coals. Both Good and VanBriesen (2019) and Cornwell et al. (2018) relied on statistics derived from coal rank data that did not take these variations into account.

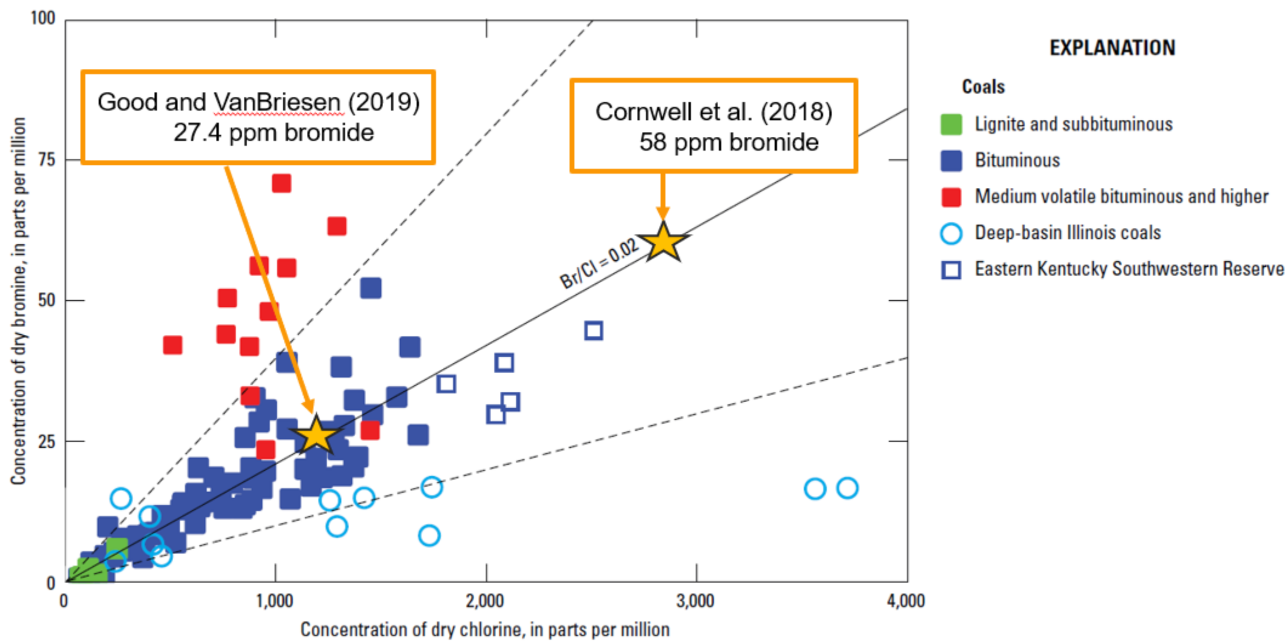

Figure 2-1
Comparison of Cornwell et al. (2018) and Good and VanBriesen (2019) bituminous coal bromide assumptions to COALQUAL data (figure adapted from Kolker et al. 2012)

Estimates of Native Coal Bromide: Impact on Article Conclusions

Use of COALQUAL data to represent as-received coals adds considerable uncertainty to the estimates of power plant discharges.

- The 75th percentile of COALQUAL data used in the Good and VanBriesen (2019) bromide mass load calculations overestimated nationwide concentrations by 58 percent for bituminous coal, 5 percent for subbituminous coal, and 50 percent for lignite, due to the use of a metric based on COALQUAL data availability rather than coal usage.
- The Cornwell et al. (2018) “maximum” bituminous coal concentration of 58 ppm used to model downstream transport is above the 90th percentile of all bituminous coal and will overestimate bromide loads for many power plants.
- Application of a chloride:bromide ratio to COALQUAL chloride data to estimate coal bromide, the approach used in Cornwell et al. (2018) and Good and VanBriesen (2016; 2017) adds uncertainty to the analysis; as illustrated in Figure 2-1, there is considerable difference in ratios among different bituminous coal sources.
- Power plants commonly source coals from multiple mines with markedly different bromide contents; neither Good and VanBriesen (2019) nor Cornwell et al. (2018) account for this practice. A more accurate method to determine inputs for a bromide mass loading calculation would be a site-specific evaluation of as-received bromide of the coals used at the plant.

3
BROMIDE ADDITION

Coal-fired power plants that do not meet mercury emission limits set by the Mercury and Air Toxics (MATS) rule using conventional pollution control devices can be brought into compliance by modifying existing equipment or by using additives. Several of the approaches involve introduction of bromide:

- Calcium bromide may be added to the power plant combustion system, either with the coal, or, less commonly, sprayed into the furnace or downstream of the furnace. Bromide added this way increases the oxidation of mercury in the flue gas so that the mercury can be removed more efficiently in air pollution control devices.
- A brominated powdered activated carbon (Br-PAC) may be injected into the flue gas downstream of the furnace. The Br-PAC adsorbs mercury from the flue gas, and is then removed in the particulate control device. The Br-PAC also increases oxidation of mercury in the flue gas, increasing mercury removal in the FGD. Br-PAC usage was not considered in any of the articles reviewed by EPRI.
- The plant may burn refined coal, which can contain added bromide. Refined coal refers to coal of any rank modified with additives to meet the conditions for a tax credit under Section 45 of the American Jobs Creation Act of 2004. For a coal producer to qualify for the refined coal tax credit, a qualified professional engineer must conduct a demonstration that burning the refined coal results in a 20% emissions reduction of nitrogen oxide and a 40% emissions reduction of either sulfur dioxide or mercury compared with the emissions that would result from burning feedstock coal. The additives used in refined coal are proprietary to the coal producer, but most refined coals contain added halogen, either calcium bromide ($CaBr_2$) or less commonly, an iodide compound, which can increase mercury oxidation and capture in FGD systems.

Bromide Addition Assumptions

Good and VanBriesen (2019; 2017; 2016) and Cornwell et al. (2018) each assumed a bromide addition rate to model downstream impacts on DBP formation.

- Good and VanBriesen (2019) applied a baseline bromide addition of 100 ppm to dry coal in modeling downstream transport for plants that added bromide in refined coal or for mercury control. For the sensitivity analysis, the paper used a low value of 10 ppm and an upper value of 460 ppm, based on a range of refined coal bromide addition rates cited by EPRI (2014a) as well as other references published prior to 2011. The sensitivity analysis for the modeling effort (Supporting Information, Figure S3) indicated that refined coal added bromide content has the greatest impact of any variable on the conclusions of the paper. Changing the bromide addition rate from 100 to 10 ppm lowers the estimated nationwide bromide mass loading from EGUs by 67 percent while changing it to 460 ppm increases the mass loading by 237 percent. Thus, an inaccurate estimate of refined coal bromide has a strong impact on the conclusions.

- Cornwell et al. (2018) assumed a worst-case addition of 300 ppm bromide to dry coal to model bromide transport in the Ohio River, based on an EPRI engineering study (Chang et al. 2010). Bromide addition was assumed for future use; Cornwell did not determine whether the modeled plants were adding bromide.
- Good and VanBriesen (2016 and 2017) assumed a minimum of 3 ppm, a most likely value of 35 ppm Br, and a maximum of 106 ppm added to dry coal as inputs to the Monte Carlo analysis of bromide loadings.

The amount of added bromide substantially drives the conclusions of both the Good and VanBriesen (2019) and Cornwell at al. (2018) evaluations. Good and VanBriesen (2019) used EIA data to identify U.S. power plants with wet FGD scrubbers that burned refined coal. Cornwell et al. (2018) modeled seven power plants discharging to the Ohio River watershed that were assumed to add bromide for MATS compliance at a future date. EPRI did not attempt to verify current bromide addition for each facility. However, of the seven plants on the Ohio River modeled by Cornwell et al. (2018), two are no longer in operation and a third does not discharge to surface water.

In addition, both articles relied on bromide addition rates reported from 2010 through 2014, several years before plants had accumulated significant operating experience with bromide addition (either as refined coal or to develop approaches for future MATS compliance). EPRI conducted four additional member surveys (EPRI, 2014b; 2015; 2016; 2017) that provide more current information. The purpose of these surveys was to evaluate potential Balance of Plant (BOP) impacts of refined coal usage such as air heater corrosion; thus, the responses were limited to power plants that were testing or using bromide for mercury control. However, EPRI attempted to obtain as broad a response as possible from power plant owners, and there was strong interest among the participants to address the observed issues with corrosion. Although the responses do not include all U.S. power plants that were testing or using bromide addition, they represent a much more robust data set than that referenced in the two articles under consideration. The survey responses indicated that after 2014, many power companies concluded that high bromide addition rates were problematic due to air heater corrosion, as well as unnecessary to meet the flue gas mercury emission standard. In response, refined coal producers obtained Section 45 certifications with lower amounts of bromide added to the coal.

EPRI reviewed bromide addition rates provided by owners of 108 coal-fired units that participated in the surveys from 2014 through 2017. The respondents included plants that used refined coal or added bromide for mercury control as part of a MATS compliance strategy. In the most recent survey (EPRI, 2017), respondents were asked if they had reduced their bromide addition rate over time. Of 63 EGUs whose owners participated in that survey, 35 units (56%) had decreased the halogen addition rate since commencing use of the technology (EPRI, 2017). Many of the originally surveyed plants have since been retired. As of December 31, 2018, survey respondents with wet FGDs operated a total of 21 power plants with 42 EGUs.

In the surveys, bromide addition rates were variously reported as ppm (mg/kg) Br in dry coal, as a weight percent of calcium bromide solution added to as-received coal, or as gallons of solution added per ton of as-received coal. EPRI converted all reported values to ppm bromide (mg/kg dry coal). Figure 3-1 illustrates the trend in bromide addition over time as cumulative distribution plots of bromide usage rates for each year of the EPRI surveys. To represent the full spectrum of bromide usage across the current coal-fired power plant fleet, Figure 3-1 includes data for all plants with wet FGDs that were still operating as of December 31, 2018, including plants that did not discharge to surface water and those that reported having ceased bromide addition over the course of the surveys (zero addition rate).

The median bromide addition rate in 2017 among all EGUs participating in the surveys was 30 ppm, compared to 90 ppm in 2016, the reference year used by Good and VanBriesen (2019). In 2016, 53% of units responding to the survey reported adding bromide at addition rates less than the 100-ppm baseline value assumed by Good and VanBriesen (2019). One year later, 77% of units responding to the survey added bromide at rates less than 100 ppm; of these units, seven (23%) had stopped adding bromide.

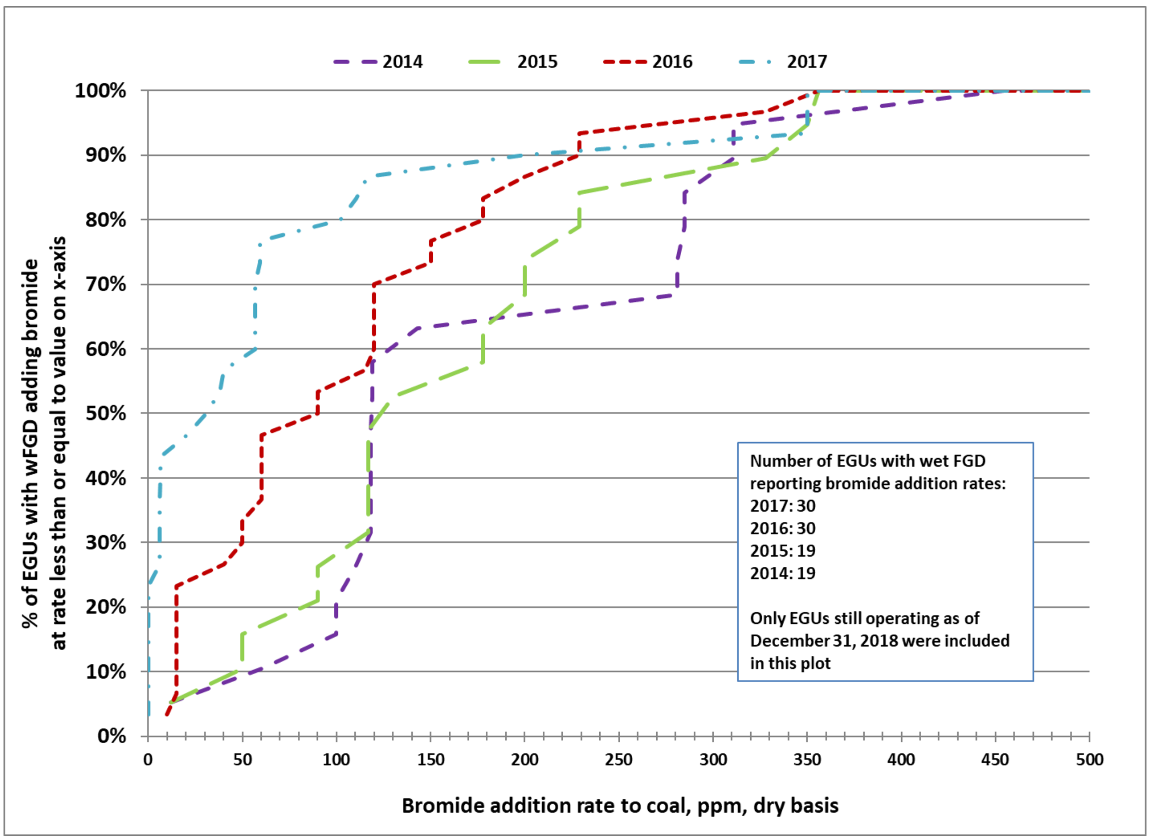

Figure 3-1
Cumulative distribution plot of bromide addition rates for all units in EPRI surveys with wet FGDs

Figure 3-2 shows the cumulative distribution plots of <u>non-zero</u> bromide addition rates from both the 2017 survey and the most recent response in any year of the surveys. The seven plants that stopped adding bromide to the coal were excluded from this plot. The median non-zero bromide addition rate in 2017 was approximately 60 ppm. Eighty-three percent of the 35 units surveyed (29 units) reported bromide addition rates of 120 ppm or less. The remaining six units added bromide at rates of 200 ppm (one unit), 230 ppm (two units at one plant), and 350 ppm (three units at one plant); these are all zero liquid discharge plants. None of the units in EPRI's 2014-2017 surveys added bromide at the 460-ppm maximum addition rate of Good and VanBriesen (2019). The 100-ppm value used in the Good and VanBriesen (2019) transport model is approximately the 75th percentile of added bromide from the 2014 through 2017 surveys. The worst-case rate of 300 ppm assumed by Cornwell et al. (2018) is five times the median of the addition rates from the 2017 survey. Based on the data collected by EPRI, a more appropriate baseline value for downstream modeling of current bromide addition would be 60 ppm.

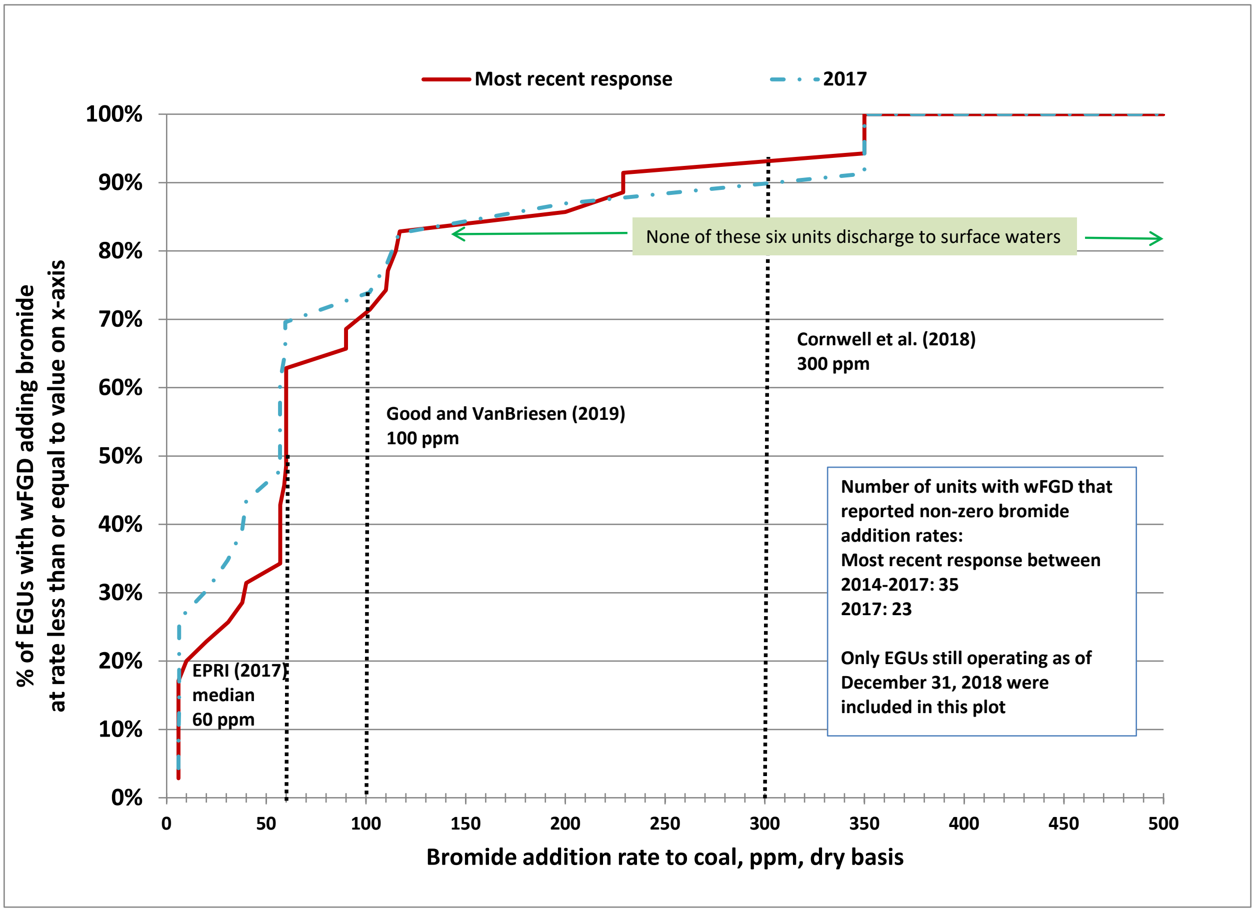

Figure 3-2
Cumulative distribution plot of survey respondents that reported adding bromide to the coal in their most recent survey response

The units included in the EPRI surveys may not be representative of all coal-fired power plants that discharge to surface water; however, EPRI's data set contains more units, with much more recent addition rate data, than in the papers that were cited by Good and VanBriesen (2019) as the basis for their addition rates. The units responding to EPRI surveys represent about 19 percent of the total population of EGUs equipped with wet FGDs.

Native Bromide Concentration for Refined Coal

Refined coal is most commonly made by adding proprietary additives to the feedstock coal at the power plant. The EIA tracks coal tonnage delivered to the plant by the rank of coal (e.g., as bituminous, subbituminous, or lignite). The EIA then tracks the firing of the coal to generate electricity as refined coal and not by coal rank. Both the amount of coal by rank delivered to the plant and the amount of refined coal fired by the plant are available in EIA Form 923, so it is possible to ascertain the rank of coal that formed the starting feedstock for the refined coal at each plant.

Good and VanBriesen (2019) used EIA Form 923 data to calculate the amount of each rank of refined coal fired at plants with wet FGDs. The authors then calculated 25th percentile, median, and 75th percentile native bromide concentrations for refined coal[3] by weighting the bromide concentration based on the 2016 industry usage of each refined coal rank. These calculated values were used to estimate bromide mass loadings to surface water for all modeled EGUs that reported burning refined coal, irrespective of actual coal rank burned. While all three concentrations were used to calculate bromide mass loadings, only the "upper" concentration was used to estimate downstream population vulnerability. Table 3-1 illustrates the approach used to calculate the 16.4 ppm "upper" value.

Table 3-1
Good and VanBriesen (2019) calculation of refined coal native bromide concentration

Refined Coal Rank	Refined Coal Use at Wet FGD Units (metric tons)	Percentage of Total 2016 Usage (weight%)	75th Percentile Bromide Concentration (ppm, dry)
Bituminous	31,555,000	50%	27.4
Subbituminous	25,320,000	40%	5.4
Lignite	6,435,000	10%	5.5
Total refined coal usage	63,310,000	100%	
Calculated "upper" concentration of native coal bromide, used for all refined coals	(0.5*27.4) + (0.4*5.4) + (0.1*5.5) = 16.4 ppm, dry		

Source of refined coal usage data and native coal bromide concentrations: Good and VanBriesen (2019) Supporting Information.

For bituminous coals, this approach produced a value closer to the median for the coal rank than to the 75th percentile (see Figure 3-3). For subbituminous and lignite coals, the approach produced values about three times higher than the 75th percentile of native bromide for that coal rank. A more accurate approach would have been to use the rank-based native bromide

---

[3] Good and VanBriesen (2019) Supporting Information, Table S-4 designates these three estimates as "lower", "middle" and "upper".

concentrations associated with the received coal when calculating wet FGD bromide loads for refined coal plants.

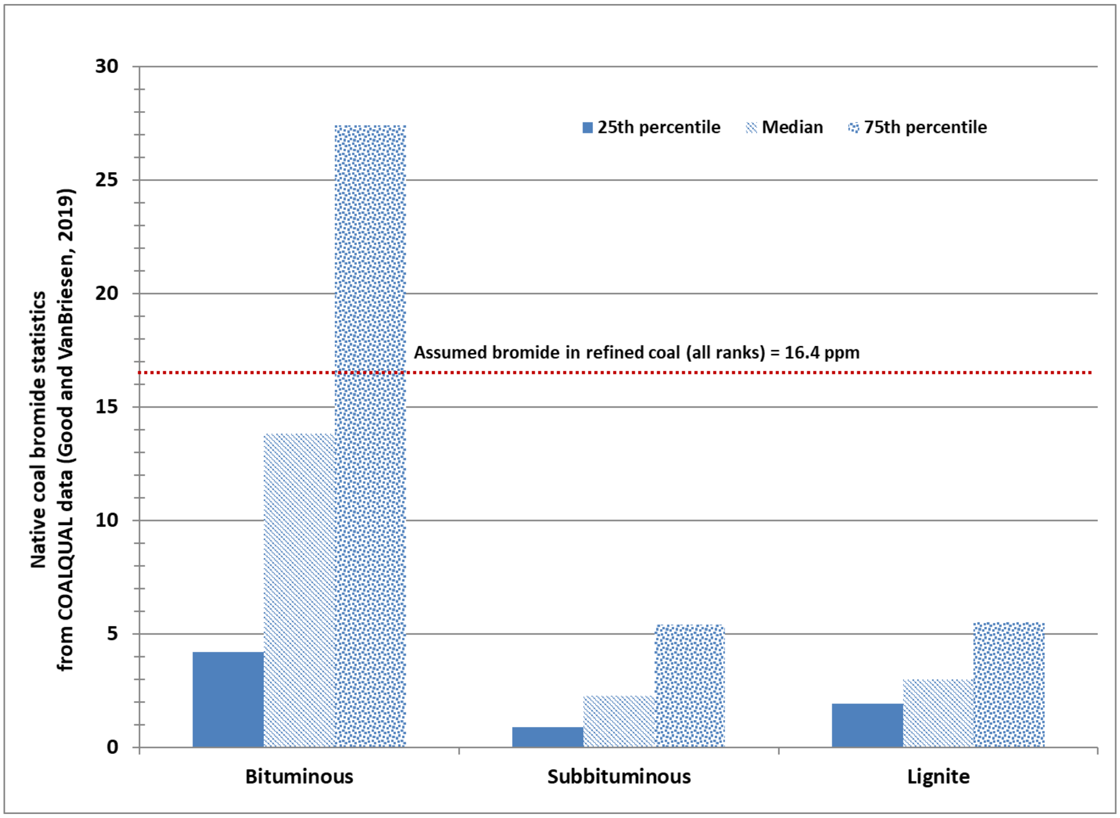

Figure 3-3
Comparison of bromide in native coal and calculated "upper" value in refined coal

Moisture Content of Refined Coal

Similar to the approach used for native bromide in refined coal, Good and VanBriesen (2019) calculated a weighted average moisture content for refined coal, based on 2016 refined coal usage of each coal rank. The resulting value of 16% moisture used in the bromide mass loading calculations does not represent any of the coal ranks accurately, overestimating moisture for bituminous and underestimating moisture for subbituminous and lignite coal. The Good and VanBriesen (2019) sensitivity analysis (Table S4) stated that the moisture content of refined coal had the fourth highest impact on model uncertainty of any variable. Use of the weighted average refined coal moisture would underestimate bromide mass loading by 10% and overestimate bromide mass loading for subbituminous and lignite coals by 15% and 27%, respectively. A more accurate approach would have been to use the actual moisture content of the refined coal feedstock.

Estimates of Bromide Addition Rate: Impact on Article Conclusions

The amount of bromide added to coal for mercury control substantially drove the conclusions of all the modeling efforts reviewed for this report. The articles relied on data from early testing of bromide addition; EPRI surveys indicate that the amount of bromide added to refined coal has decreased significantly in recent years.

- The 100-ppm addition rate assumed by Good and VanBriesen (2019) is at the high end of rates reported for all units that discharge to surface water in the most recent EPRI survey. Owners of six units reported using higher rates, but those are all zero liquid discharge plants. The median addition rate for all FGD plants still adding bromide in their last survey response was 60 ppm. Thus, the 100-ppm assumption likely overestimates downstream impacts for most facilities.
- The 300-ppm addition rate assumed by Cornwell et al. (2018) for modeling potential future impacts on downstream drinking water sources is nearly three times the highest rate reported in the most recent EPRI survey for units that discharge to surface water. Due to this assumption, the model would have significantly overestimated the magnitude of any future downstream bromide impacts.
- Weighted average native coal bromide and moisture values used in the Good and VanBriesen (2019) model for plants that burned refined coal reduce the accuracy of the bromide mass loading estimates.

4
PARTITIONING OF BROMIDE TO FGD WASTEWATER

When coal is fired in a power plant furnace, the bromide associated with that coal (whether native or added) can exit the plant in multiple waste streams, including flue gas, combustion byproducts (bottom ash and fly ash), or in FGD wastewater or solids (if a wet FGD scrubber is present). A schematic of a typical coal-fired power plant with an FGD system is shown in Figure 4-1. The most important pieces of equipment for the fate of bromide are the furnace, the fly ash collection system, the FGD system (if one is present) and the stack.

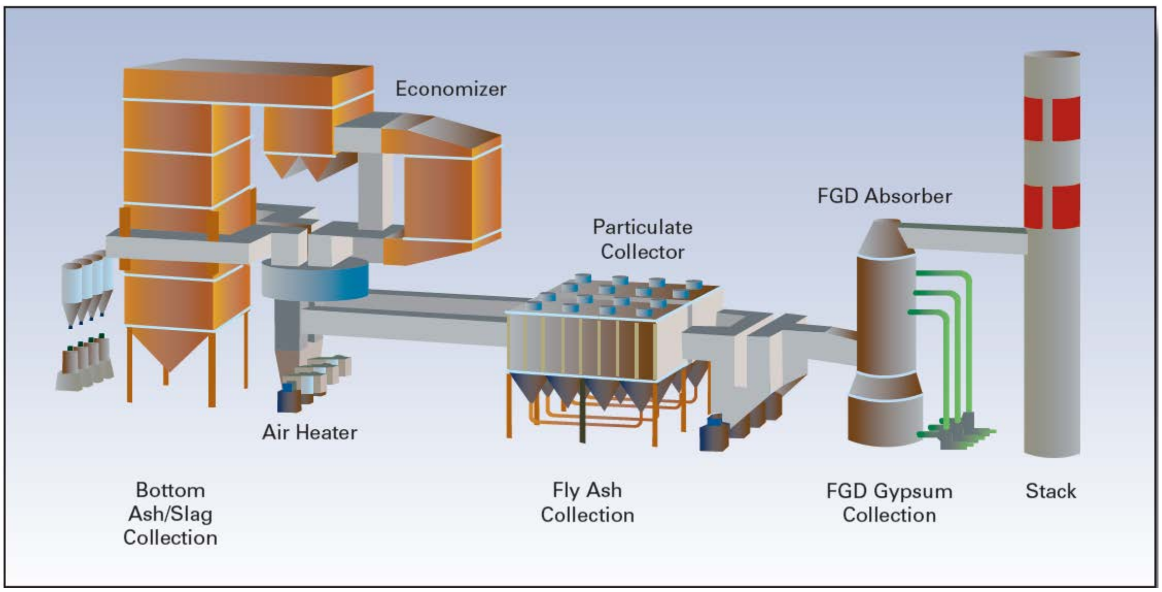

Figure 4-1
Schematic of a typical coal-fired power plant

EPRI (2014a) reviewed two published studies that attempted to establish mass balances for bromine in plants with a wet FGD by measuring concentrations in the coal, flue gas, and various waste streams (Buschmann et al., 2005; EPRI, 2009). However, only one study (Buschmann et al., 2005) evaluated partitioning between FGD solids and liquids at a single power plant, reporting that 82 percent of coal bromide ended up in the FGD liquid and less than one percent in the FGD solids and gypsum. The remainder of the bromide was adsorbed to fly ash (12 percent) and emitted in stack gas (four percent). EPRI (2009) used flue gas measurements to determine the removal of bromide from flue gas in a single pilot FGD system; the study did not determine the bromide content of FGD wastewater. The study reported that 70 to 90 percent of the bromide in coal was found in the flue gas entering the FGD system, and that 94 to 97 percent of the bromide in the flue gas was removed in the scrubber.

In modeling downstream bromide impacts, Good and VanBriesen (2019) used a value for bromide partitioning from coal to FGD wastewater of 84%, based on a literature search reported in Good and VanBriesen (2016). Cornwell et al. (2018) assumed that 90% of bromide in coal is found in the wet FGD liquid, citing two of the same studies cited by Good and VanBriesen (2016). Of these two studies, one (Srivastava et al., 2006), does not discuss bromide removal in FGDs at all; the article is a general discussion of the efficacy of controls for mercury removal. The other study cited (Meij, 1994) reports high removal of bromide from flue gas by an FGD system but does not provide a mass balance.

Bromide Partitioning: Impact on Article Conclusions

There is insufficient published information to suggest an appropriate value for bromide partitioning to FGD wastewater. There are few published studies and the plants studied are not representative of all power plant systems. However, the values used in the models are within the bounds of the limited data that do exist, and the assumptions made by the authors are not likely to have a major impact on the model conclusions compared to the other uncertainties in the downstream impact modeling.

5

NUMBER OF FGD WASTEWATER DISCHARGES

Good and VanBriesen (2019) identified 140 coal-fired power plants with wet FGDs, of which 116 were assumed to discharge to surface water. The list was compiled from 2016 EIA records and supporting information from the US EPA's Effluent Limitation Guidelines rulemaking. Facilities that discharged to the Great Lakes, estuaries or bays, and facilities that disposed of FGD wastewater via deep well injection or evaporation ponds were excluded from the count. Plants with planned zero liquid discharge (ZLD) processes or with unclear ZLD status were retained in the list. The exact rationale for including or excluding each plant was not presented. The count of discharging plants was used to map cumulative downstream bromide concentrations at downstream drinking water intakes and to develop a map of vulnerability to populations served by those water utilities. Thus, the list of operable plants discharging to surface water is a key factor in the Good and VanBriesen (2019) analysis.

Recent years have seen a rapid decline in the number of coal-fired power plants as older facilities are shut down and replaced with gas generation and other power sources. Using the most recent reliable information is essential to evaluate the prevalence of potential downstream impacts of bromide from power plant discharges. EPRI maintains an ongoing list of all U.S. power plant EGUs greater than 50 MW capacity, their current operating status, whether they have wet FGD systems, and whether they discharge FGD wastewater to surface water. For the following discussion, EPRI included EGUs operational as of December 31, 2018. Units with announced closures after that date were retained in our analysis.

EPRI compared the Good and VanBriesen (2019) count with a count generated by EPRI from a comprehensive list of coal-fired power plant stations and units developed by James Marchetti, Inc. The primary data source or foundation of the Marchetti list is the Emission-Economic Modeling System (EEMS) Database. The primary information sources used to maintain the database are Energy Information Administration (EIA) Forms 860 and 923, published reports, and individual discussions with electric generators. The characteristics of each power plant unit (e.g., current or proposed air pollution control device, particulate emissions) and the FGD wastewater discharge status of each facility is also included in the list.

EPRI reviewed the discharge locations and surface water classifications of power plants located on the Great Lakes, estuaries and bays and removed seven facilities that discharge to those water bodies from consideration. EPRI also verified the discharge status of each power plant identified as potentially ZLD in Good and VanBriesen (2019, Supporting Information Table S2): all were confirmed to not discharge to surface water and were removed from our count.

EPRI identified 87 plants operational as of December 31, 2018 with at least one unit with a wet FGD that discharge FGD wastewater to surface water outside of the Great Lakes, bays and estuaries. There were 220 coal-fired EGUs associated with these plants, representing 113 gigawatt (GW) of generating capacity. Table 5-1 summarizes these findings, which indicate that the number of plants that could potentially impact downstream drinking water is 25 percent fewer than modeled by Good and VanBriesen (2019). The decrease is due to retirements and verification by EPRI of ZLD status.

Table 5-1
Good and VanBriesen (2019) and EPRI tallies of plants with wet FGDs discharging to surface water

	Good and VanBriesen, 2019 (based on 2016 EIA data)	**EPRI (based on EGU status as of 12/31/18)**
No. of plants with at least one operable wet FGD system discharging to surface water[1]	116	87 (based on unique Facility IDs)

[1] Excluding plants that discharge to the Great Lakes, bays and estuaries.

Table 5-2 lists the nine plants that were modeled in Good and VanBriesen (2016 and 2017). Seven plants were operating with FGD discharge in December 2018; Harrison has not discharged to surface water since 1995 and Longview has never discharged to surface water. Table 5-3 lists the plants on the Ohio River evaluated in Cornwell et al. (2018); only four of the seven plants were operational in 2018 and discharging to surface water. One (Cane Run) is retired, one (Coleman) is permanently idled, and one (East Bend) does not discharge to surface water.

Table 5-2
Plants with wet FGDs modeled in Good and VanBriesen (2016, 2017)

Plant Name (State)	Plant ID	Units with Wet FGD Operational on Dec. 31, 2018	Discharge to Surface Water?
Cheswick (PA)	8226	1	Yes
Keystone (PA)	3136	2	Yes
Homer City (PA)	3122	1	Yes
Conemaugh (PA)	3118	2	Yes
Brunner Island (PA)	1616	3	Yes
Fort Martin (WV)	1152	2	Yes
Longview (WV)	808	1	No
Montour (PA)	1775	2	Yes
Harrison (WV)	2052	None	No

Table 5-3
Plants on the Ohio River referenced in Cornwell et al. (2018) with wet FGDs

Plant Name (State)	Plant ID	Units with Wet FGD Operational on Dec. 31, 2018	FGD Discharges to Surface Water?
Cane Run (KY)	1363	Retired	No
East Bend (KY)	6018	1	No
Clifty Creek (IN)	983	6	Yes
Mill Creek (KY)	1364	4	Yes
Coleman (IN)	1381	0[1]	No
F. B. Culley (IN)	1012	2	Yes
A. B. Brown (IN)	6137	2	Yes

[1] Units have been idled since 2014 and are shut down indefinitely.

Power Plant and Unit Counts: Impacts on Article Conclusions

The articles reviewed by EPRI used information for specific coal-fired power plants to draw conclusions on downstream impacts of bromide. Due to recent plant closings and changes to plant water handling practices, the information in these articles is out of date. Information gathered by EPRI, current as of December 31, 2018, indicates that both Good and VanBriesen (2019; 2017; 2016) and Cornwell et al. (2018) overestimate the number of plants that could potentially impact downstream surface water.

- Of the 116 coal-fired power plants evaluated by Good and VanBriesen (2019) based on 2016 EIA data, EPRI tallied 87 that are were still in operation as of December 31, 2018 and discharged FGD wastewater to surface water (not including the Great Lakes and estuaries). This results in 25% fewer power plants discharging to surface water compared to the Good and VanBriesen (2019) analysis.
- Of the nine facilities modeled by Good and VanBriesen (2016; 2017), one is no longer in operation and another does not discharge to surface water.
- Of the seven plants on the Ohio River modeled by Cornwell et al. (2018), two are no longer in operation and one does not discharge to surface water; thus, the impacts identified in the report do not represent current conditions.

6
BROMIDE MASS LOADS TO SURFACE WATER

This section reviews the procedures used by Good and VanBriesen (2016; 2017; 2019) and Cornwell et al. (2018) to estimate bromide mass loadings from power plants to surface water.

Good and VanBriesen (2019)

Good and VanBriesen (2019) developed three estimates of bromide mass loading to surface water, differing only in coal native bromide concentration. The three estimates used identical values for bromide addition (100 ppm, dry), coal moisture, and partitioning of bromide from flue gas into FGD wastewater (84%). For EGUs that did not burn refined coal, the following values were used in the three estimates:

- The "Low," estimate used the 25th percentile of coal bromide data in the COALQUAL database for each coal rank
- The "Mid," estimate used the median of COALQUAL data for each coal rank
- The "High," estimate used the 75th percentile of COALQUAL data for each coal rank

For EGUs that were determined to add bromide or burn refined coal, the three estimates used the calculated Low, Middle, and Upper refined coal values listed in Table S4 of the Supporting Information, regardless of the rank of the coal feedstock, as discussed in Section 3.

While all three bromide mass loading values were used in downstream transport modeling (Good and VanBriesen, 2019 Supporting Information), EPRI focused on the "High" estimate, because that set of values was used to determine potential downstream population impact.

Good and VanBriesen (2016; 2017)

Good and VanBriesen (2016; 2017), which evaluated potential downstream impacts from specific power plants on the Allegheny River and in watersheds in Pennsylvania, respectively, used a different methodology to calculate bromide loadings than Good and VanBriesen (2019). Native coal bromide concentrations were estimated by applying a ratio of 0.02 to the COALQUAL chloride concentration statistics. However, a Monte Carlo analysis was then used to estimate bromide mass loadings to surface water. Parameters varied in the analysis were coal bromide concentration, coal moisture, coal usage, bromide addition rate, and bromide capture in the FGD. Because the model did not use discrete values of each of these parameters directly in the transport model but rather the quartiles of the Monte Carlo output, they cannot be compared directly to Good and VanBriesen (2019).

Cornwell et al (2018)

Cornwell et al. (2018) estimated maximum potential bromide mass loadings to the Ohio River using a high bituminous coal bromide concentration of 58 ppm, a maximum added bromide concentration of 300 ppm (assumed to apply to all plants evaluated whether they added bromide or not), and an average moisture content of 6.5%. The article does not provide details of the method used to calculate the bromide mass loading from each facility, but states that it follows the methodology of Good and VanBriesen (2016).

Error in Calculation of Dry Coal Tonnage

To calculate the bromide mass loading from each EGU, Good and VanBriesen (2019) converted as-received (wet) coal weight to dry coal weight, so that the coal amount could be combined with the bromide concentration in ppm, dry. Equation (1) from the article is shown below.

$$\begin{pmatrix}\text{Estimated}\\ \text{wet FGD}\\ \text{Br load,}\\ \text{kg/day}\end{pmatrix} = \begin{pmatrix}\text{wet FGD}\\ \text{coal}\\ \text{consumption,}\\ \text{as received,}\\ \text{million kg/day}\end{pmatrix} \times \left(\frac{1}{1\text{-}\begin{pmatrix}\text{moisture}\\ \text{content, \%}\end{pmatrix}}\right) \times \begin{pmatrix}\text{Br capture}\\ \text{in wet}\\ \text{FGD, \%}\end{pmatrix} \times \begin{pmatrix}\text{Br}\\ \text{content}\\ \text{dry coal,}\\ \text{ppm}\end{pmatrix} \text{(Eq. 1)}$$

This equation has two errors. First, (1-moisture content) should be in the numerator, not the denominator. Second, the equation as written should use a moisture content fraction, not a percent. A corrected equation is shown below, with expanded units shown for clarity. The wet coal amount is shown in kilograms per day (kg/day) rather than million kg/day, and the bromide concentration is shown in milligrams per kg (mg/kg, equivalent to ppm). These changes were made to provide better clarity on the unit conversions.

$$\begin{pmatrix}Estimated\\ wet\ FGD\\ Br\ load,\\ kg/day\end{pmatrix} = \frac{wet\ FGD\ coal\ consumption, kg\ as\ recd\ coal}{day} \times \frac{(1 - moisture\ fraction)\ kg\ dry\ coal}{kg\ as\ received\ coal} \times \frac{Br\ capture\ in\ wet\ FGD, \%}{100} \times \frac{Coal\ Br\ content, mg}{kg\ dry\ coal} \times \frac{kg\ Br}{10^6\ mg\ Br}$$

Assuming that the Good and VanBriesen (2019) calculations did in fact use a moisture fraction, the moisture conversion error would result in a substantial overestimate of dry coal tonnage and bromide mass loading for each EGU: by 14% for bituminous coal, 88% for subbituminous coal, 130% for lignite coal, and 42% for refined coal. Table 6-1 illustrates the impact of the error on the "upper" bromide mass loading, for an EGU burning one million kg per day of bituminous coal or refined coal.

Good and VanBriesen (2016) used a variant of the same equation to calculate bromide mass loading inputs to the Monte Carlo estimation method. Good and VanBriesen (2017) references the 2016 methodology. Thus, it appears that all three publications may suffer from the moisture conversion calculation error. The extent to which downstream bromide concentrations from individual power plants were impacted by the error cannot be determined from data presented in the article and Supporting Information. Because bromide mass loadings were calculated for each EGU independently, different power plants could be impacted to a different extent depending on which coal ranks were burned in the individual EGUs.

Table 6-1
Impact of coal moisture error on bromide mass loadings: bituminous coal and refined coal

Equation Source	Wet Coal (kg/day)	Moisture fraction	Wet coal/dry coal conversion factor[2]	Br capture in wet FGD	Br content, mg/kg (ppm), dry	Br mass, kg/day	Difference in kg Br/day
Bituminous coal							
G&VB[1]	1,000,000	0.065	1.07	84%	27.4	24.6	14.4%
Corrected	1,000,000	0.065	0.935	84%	27.4	21.5	
Refined coal							
G&VB[1]	1,000,000	0.16	1.19	84%	16.4	16.4	41.7%
Corrected	1,000,000	0.16	0.84	84%	16.4	11.6	

[1] Good and VanBriesen (2019) Equation 1
[2] The wet coal to dry coal conversion factor is the result of the second term of each of the equations shown above.

The moisture conversion errors in Good and VanBriesen (2019) can be combined with the bias from the use of the 75th percentile of COALQUAL sample count-weighted bromide concentrations (Section 2). For a power plant burning only bituminous coal, the combined bias produces an 80% overestimate of bromide mass loading to water [(1.14 * 1.58) - 1= 0.8]. For subbituminous and lignite coal, the overestimates are 98% and 245%, respectively. The impact on plants burning refined coal will depend on which coal type was used as the feedstock, but the result will be to overestimate bromide mass loadings in all cases.

Good and VanBriesen (2019) Range of Load Estimates Varies Among Regions

Good and VanBriesen (2019) Table S-12, Supporting Information lists three estimates of total bromide mass loadings, by U.S. hydrologic region. The low, mid, and high loading estimates use the 25th, median, and 75th percentiles of native coal concentration, respectively. Figure 6-1 displays ratios of the “high” load estimates to the “mid” level estimates (shown in orange), and ratios of the “high” versus “low” load estimates (shown in gray) for each of the hydrologic regions. On average, the high load estimate is 1.6 times the mid-level estimate, and 3.4 times the low estimate. However, in some regions, the high estimate is 6.5 times the low estimate. These differences are due to different usage of native coal ranks and refined coal among the regions. Because of the substantial difference in these estimated loads, relying on only the high loading estimate to assess downstream impact may overestimate potential impacts on downstream water supplies.

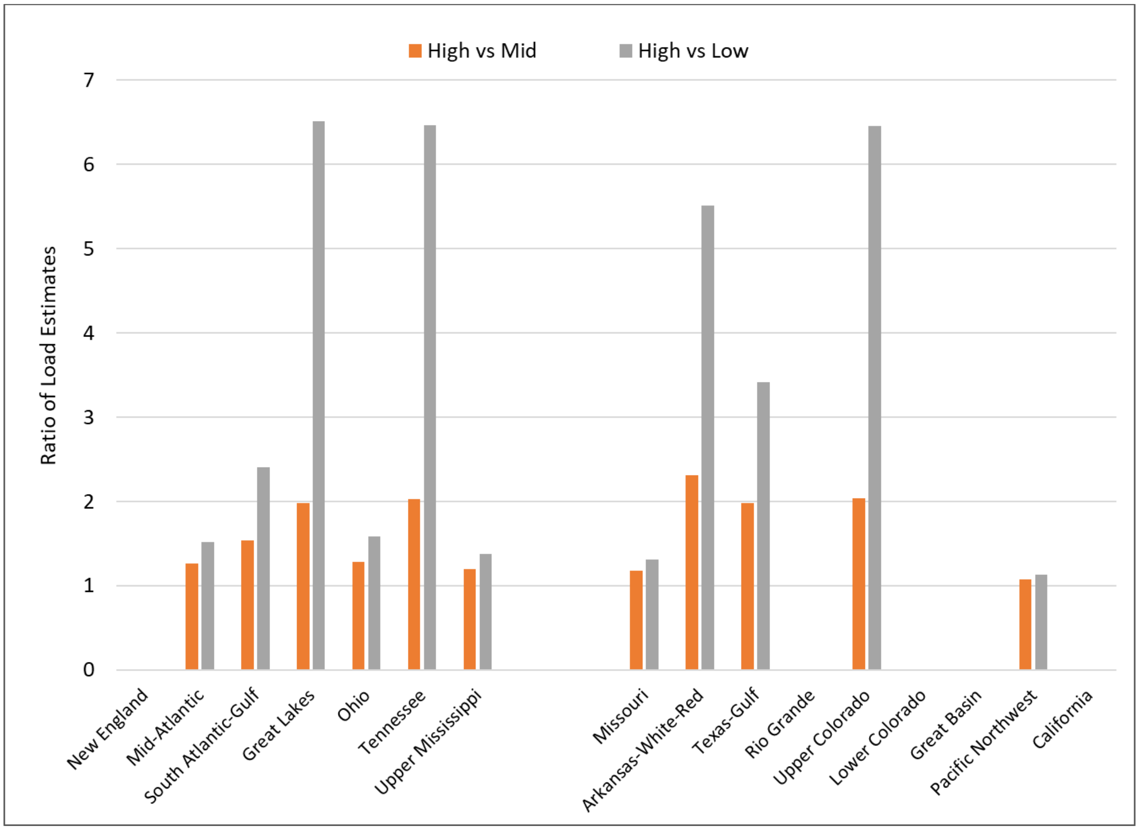

Figure 6-1
Good and VanBriesen (2019) modeled bromide load estimates by hydrologic region

Estimates of Bromide Mass Loads: Impact on Article Conclusions

The bromide mass load calculations incorporate all the uncertainties and biases discussed above; thus, the accuracy of the estimates depends on the validity of the individual values used in the models.

- The bromide mass loadings in all three Good and VanBriesen articles are called into question due to an error in the equation used to calculate the amount of dry coal burned in the modeled power plants. If the equations were used as presented, they would produce overestimates of bromide mass loadings ranging from 14% for EGUs burning bituminous coal, to 130% for EGUs burning lignite coal. EPRI received confirmation from Dr. VanBriesen that the 2016 article equation is incorrect, but the response did not address the other two articles or state whether the results of the models were impacted (VanBriesen, 2019). EPRI will continue to investigate the extent to which the erroneous equation has impacted the conclusions of these articles.
- In Good and VanBriesen (2019), the coal moisture error can be combined with overestimates ranging from 5.5% to 58% from procedures used to estimate native coal bromide concentrations (Section 2). For power plants burning only a single, non-refined coal rank, the

combined impact on the “High” case used for downstream population vulnerability modeling is to overestimate mass loadings by 80 to 245 percent.

- Cornwell et al. (2018) used a “maximum” native bromide concentration in bituminous coal and a very high value for bromide addition to estimate potential future bromide concentrations at downstream DWTS on the Ohio River. These combined assumptions will overestimate bromide mass loads in almost all cases. In addition, it is unknown whether Cornwell et al. (2018) is impacted by the same coal moisture conversion error as the Good and VanBriesen (2016) article that is cited as the source of the methodology. If so, this will result in a significant overestimate of bromide mass loadings.

7 HYDROLOGIC MODELING

An important consideration in the assessment of the potential impacts of bromide discharges from power plants is mixing and dilution in the receiving waters, particularly given that bromide is conservative in nature (e.g., does not sorb to sediments, is not readily bioavailable, is not volatile, etc.). Thus, dilution is a primary factor that affects instream bromide concentrations. Both Good and VanBriesen (2019) and Cornwell et al. (2018) focused primarily on riverine systems. River flows typically exhibit wide variation due to rainfall, regulation by dams, and other factors. Figure 8-1 provides an example of flow variability for a randomly chosen river illustrating that daily flows varied over the course of the year by two orders of magnitude. These variations are important because an effluent load delivered to a river under low flow conditions will yield a higher concentration instream than the same load that occurs under high river flow conditions.

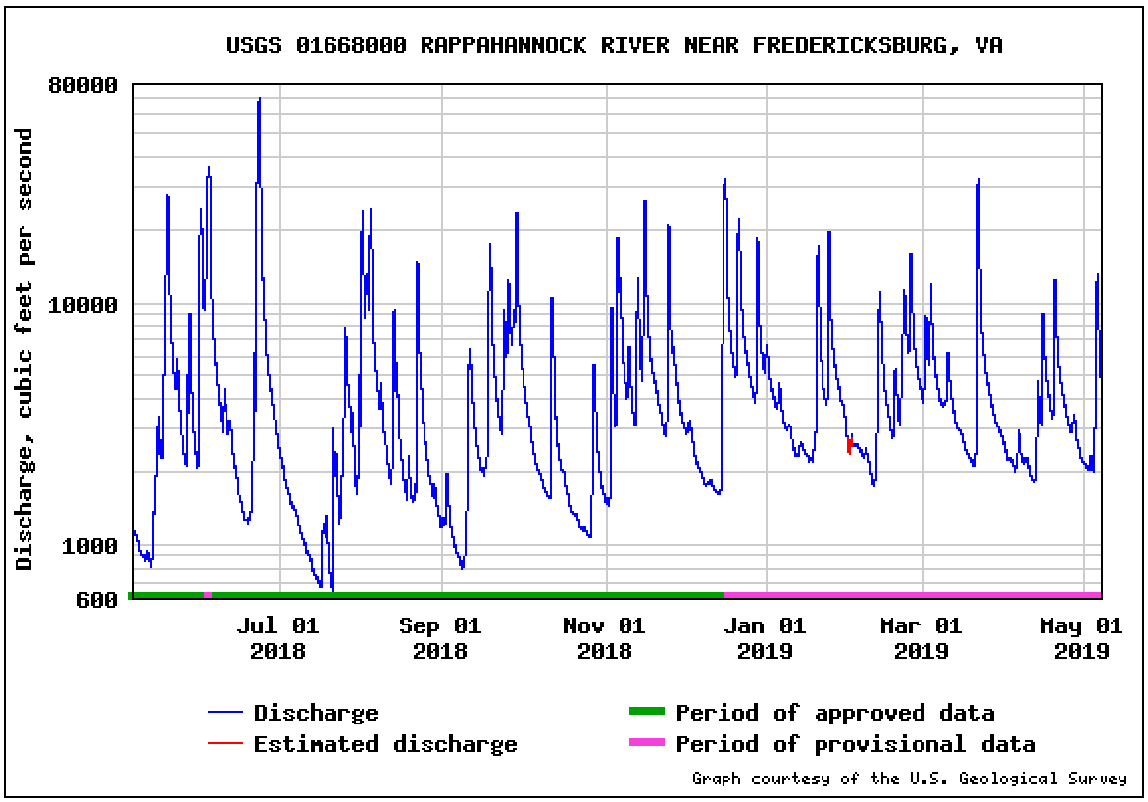

Figure 7-1
Example stream flow plot

The Good and VanBriesen (2019) downstream transport modeling was conducted using mean annual flow conditions, which does not reflect the full range of flows, and therefore, the full range of potential instream bromide concentrations. However, some type of averaging is likely appropriate, given that the potential human health concerns related to DBPs are based on long-term exposure. The study calculated bromide contributions at DWTS from upstream sources using a simple tracking approach that followed each discharge load downstream and computed concentrations in response to changes in stream flow. The transport modeling was continued downstream until the bromide contribution from a source fell below 1 microgram per liter (1 µg/L), leading to very long streamflow paths that are difficult to model accurately. To calculate an "effect metric" (the product of the surface drinking water facility population served and the wet FGD Br concentration contribution from a power plant), a mean annual flow rate was used for large reservoirs as well as rivers. This simplistic approach does not consider additional dilution due to tributary inputs or reservoir circulation (some FGD wastewaters are discharged to reservoirs with complex hydraulics), and could have produced overly conservative results.

Cornwell et al. (2018) utilized a dynamic water quality modeling approach that considered variability in flows over an entire calendar year (2014) for the Ohio and Dan rivers. However, USGS data indicate that river flows in these two rivers exceed those called out in Figures 4 through 7 of Cornwell et al. (2018) 80 percent of the time, as shown in Figures 8-2 and 8-3 below. That is, for a given bromide effluent load, estimated instream bromide concentrations would be lower than those shown in Cornwell et al. (2018) approximately 80 percent of the time. By highlighting the low flow periods, the paper very likely overestimated potential bromide impacts. A more rigorous evaluation of hydrology and hydraulics would provide a better characterization of potential instream bromide levels due to FGD discharges.

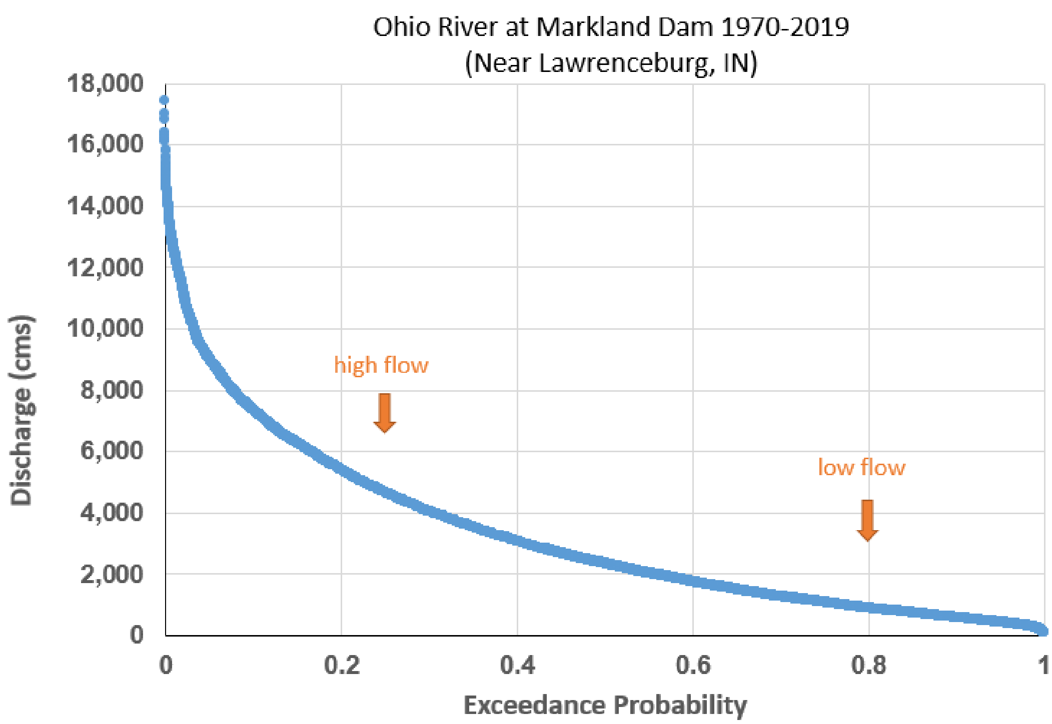

Figure 7-2
Flow distribution for the Ohio River, compared to conditions highlighted by Cornwell et al. (2018)

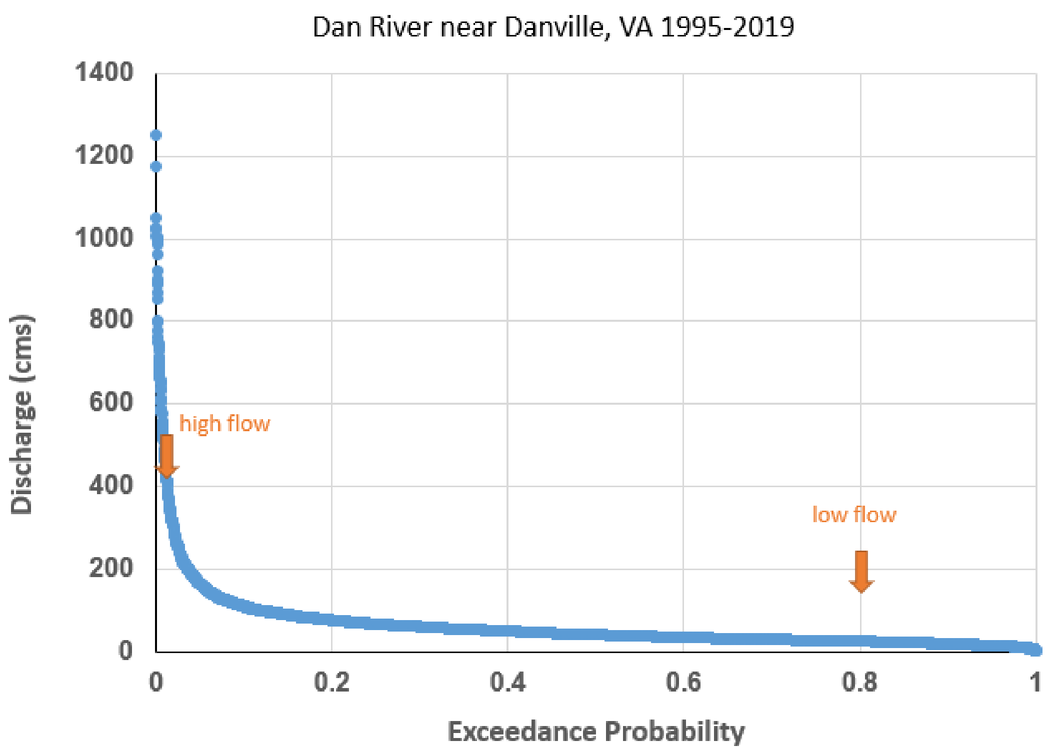

Figure 7-3
Flow distribution for the Dan River, compared to conditions highlighted by Cornwell et al. (2018)

Good and VanBriesen (2016) used a mass-balance approach to model bromide contributions from various sources, including anthropogenic (wet FGD and oil and gas wastewater discharges) and natural sources. Estimated bromide loads were modeled over a range of river flow conditions, including 25^{th} percentile, mean, and 75^{th} percentile, to evaluate the extent of potential FGD contributions. Good and VanBriesen (2017) used a similar mass-balance approach to evaluate bromide concentrations at average annual stream flows and at average August flows to represent low-flow periods.

Model Validation

Good and VanBriesen (2019) and Cornwell et al. (2018) both used estimated FGD bromide mass loadings and hydrologic modeling to forecast downstream concentrations and potential impacts. Neither paper corroborated the model results with instream data. Model verification, the process of evaluating model calculations compared to observed data to verify that the calculations are accurate, is an essential component of any modeling evaluation.

Cornwell et al. (2018) included a few general statements comparing the authors' results to data. For example, "At one WTP intake on the Dan River, the modeled bromide result was 335 µg/L, and the historical maximum concentration measured at this WTP was in the upper 400 µg/L range." This type of comparison is not helpful, since it compared the modeled result only to a maximum observed concentration and did not provide information needed to put the measurement in context (range, dates, river flows, etc.). At this DWTS intake (Figure 7 of the paper) the modeling results predicted a range of bromide concentrations from approximately 40 μg/L at "high flow" to 335 μg/L at "low flow".

Model verification by means of sampling and analysis is needed to validate the model assumptions and methodology. Without this, model results cannot be evaluated with any level of confidence. For example, Figure 2 of Cornwell et al. (2018) showed much higher bromide concentrations at a downstream water treatment plant than at an upstream location, yet there are no apparent bromide inputs between the two locations. The paper explained this discrepancy as due to the dynamic nature of the model; the higher downstream bromide is attributed to higher bromide inputs from upstream power plants on preceding days. While this may be a plausible explanation, it is impossible to assess without a comparison to actual data collected at multiple points along the river. A more robust modeling approach would include comparisons to observed instream data to provide confidence in model results and potential impacts.

The Ohio River Valley Water Sanitation Commission (ORSANCO) conducts bimonthly sampling at various locations on the Ohio River, including within the reach modeled by Cornwell et al. (2018). While the data are somewhat limited, 2014 bromide concentrations for Ohio River locations between Ohio River Mile 531.5 and Ohio River Mile 846 [corresponding to the river reach and time period modeled by Cornwell et al. (2018)] ranged from below detection to 95 μg/L (N = 27; ORSANCO, 2019). In an analysis of a different dataset from 2011-2012, ORSANCO (2014) indicated that typical bromide levels were less than 100 μg/L throughout the river. By comparison, the Cornwell et al. (2018) modeling effort, that included only power plant contributions and did not factor in background bromide, resulted in downstream concentrations greater than 120 μg/L at many locations, and greater than 200 μg/L at some locations. This suggests that the modeling results may be overestimating downstream contributions from power plants and a more robust evaluation would be appropriate.

Hydrologic Modeling: Impact on Article Conclusions

When modeling downstream impacts of a pollutant on riverine and reservoir concentrations, two very important considerations for a robust approach are flow dynamics and model verification.

- Good and VanBriesen (2019) used mean annual flow conditions which does not reflect the full range of flows, and therefore, the full range of potential instream bromide concentrations.
- Good and VanBriesen (2019) used a simplistic approach to model bromide concentrations to a very small level over long streamflow paths which did not properly consider additional dilution due to tributary inputs or reservoir circulation, likely producing overly conservative results.
- Although Cornwell et al. (2018) considered variability in flows over an entire calendar year, USGS data indicated that actual river flows exceeded the low flow values the authors focused on 80% of the time, very likely resulting in overestimates of potential bromide impacts.

- Neither Good and VanBriesen (2019), nor Cornwell et al. (2018) verified model results with instream data. Additionally, a cursory comparison with a limited observed dataset indicates that modeled concentrations by Cornwell et al. (2018) may be greatly overestimated.

8
HEALTH RISK ASSESSMENT

The introduction of drinking water disinfection led to the greatest decrease in water-borne illnesses and deaths in human history. However, it has been known for many decades that disinfection chemicals, particularly chlorine and to a lesser extent, chloramines, can react with natural organic matter to form halogenated byproducts that have adverse human health effects. EPRI (2014a) reviewed the extensive literature on the current understanding of human health effects of both regulated and unregulated halogenated DBPs.

The human health effects most often cited as potentially associated with chronic exposure to low doses of THMs are reproductive effects, hepatic effects and bladder cancer; (USEPA, 2019a) the latter is the basis for the drinking water Maximum Contaminant Level (MCL).

- Chloroform is considered *unlikely to be a human carcinogen*, except under high-exposure conditions that lead to cytotoxicity and regenerative hyperplasia in susceptible tissues.
- Bromoform and bromodichloromethane are considered probable human carcinogens (B2), based on inadequate human data and sufficient evidence of carcinogenicity in animals.
- Dibromochloromethane is considered a possible human carcinogen (C) based on inadequate human data and limited evidence of carcinogenicity in animals.

Only two of the five regulated haloacetic acid species, dichloroacetic acid (Cl_2AA) and trichloroacetic acid (Cl_3AA), have cancer assessments in the EPA IRIS database (USEPA, 2019a): The basis for the HAA5 cancer risk factors is hepatocellular carcinoma (liver cancer). The conclusions from these assessments are as follows:

- EPA assessed Cl_2AA as *"likely to cause cancer in humans"* based on a statistically significant and dose-related incidence of hepatocellular adenomas and carcinomas in two non-human species (rats and mice). The assessment noted major uncertainties related to mode of cancer action, extrapolation of findings to humans, and extrapolation to low doses using a linear model. Some studies indicate that Cl_2AA is genotoxic, but only at very high doses unlikely to be found in drinking water.
- Cl_3AA was assessed "*there is suggestive but not conclusive evidence that Cl_3AA will cause cancer in humans"*, based on several studies showing increased liver cancer in one strain of male mice, but not in female mice. No studies on other species or on humans were available. Studies of genotoxic activity of Cl_3AA were inconclusive.

Of the articles reviewed in this report, only Good and VanBriesen (2019) made use of DBP health risk information, and only indirectly, to determine the power plant bromide contributions at a downstream DWTS at which transport modeling was extended no further downstream.

Cornwell et al. (2018) and Good and VanBriesen (2016, 2017) modeled only downstream bromide concentrations and did not evaluate human health risk associated with DBP exposures. Good and VanBriesen (2019) limited the furthest downstream extent of transport modeling based on an article by Regli et al. (2015) that identified concentrations of concern for bromide in drinking water. The authors calculated a statistical relationship between increased bromide concentrations in drinking water and increased TTHM in chlorinated drinking water and then related the increases in TTHM to increases in bladder cancer. Regli et al. (2015) concluded that an increase of 10 μg/L bromide in the inlet to a DWTS is the lowest point at which a significant increase in TTHM would be observed.

The Regli et al. (2015) risk assessment has two components:

- Changes in TTHM with increases in bromide were estimated using the EPA's Surface Water Analytical Tool (SWAT), an empirical model of disinfection processes and DBP measurements used by the EPA in the development of the Stage 2 Disinfectants and Disinfection Byproducts Rule. On a plant-month basis, the SWAT data indicated that a 10 μg/L increase in bromide was associated with a mean 1.3 μg/L increase in TTHM.

- The additional risk of bladder cancer associated with a 1 μg/L increase in TTHM on a plant-month basis was calculated to be 0.0001 or 10^{-4}, based on a pooled analysis of data from six epidemiological studies (Villanueva et al., 2004) that related residential concentration data to bladder cancer incidence.

Regli et al. (2015) combined the results of these two evaluation steps and converted them to an annual average basis, concluding that with a 50 μg/L increase in bromide above baseline levels, 90% of 201 drinking water plants in the SWAT database would have seen an increase in TTHM at or above 1 μg/L, and consequently an increased cancer risk of at least 10^{-4}. A 100 μg/L increase in bromide above baseline corresponded with an estimated additional cancer risk of 10^{-3}. However, the authors noted a diminishing rate of increase in TTHM per μg/L increase in bromide at higher initial water bromide concentrations, possibly due to limited organic precursors available for DBP formation. Thus, there is no single relationship that can be drawn between a change in bromide mass loading and the resulting level of TTHM formed in the wastewater.

EPRI reviewed the methodology and sources of the TTHM epidemiological data cited by Regli et al. (2015) and compared those to the avoided cancers analysis performed by the EPA in Appendix E to the Economic Analysis for the Stage 2 Disinfection and Disinfection Byproducts rulemaking (USEPA, 2005). A detailed evaluation of the epidemiological studies cited by Regli et al. (2015) and USEPA (2005) was outside the scope of this literature review. The calculations used by Regli et al. (2015) to relate TTHM exposures to excess lifetime cancer risk adheres to the methodology used by EPA in developing the Rule. As shown in Table 8-1, the primary epidemiology studies cited by EPA and Regli differ, as do the exposure metrics employed, but the conclusions of the two studies are similar: EPA (2005) concluded that exposure to TTHM was responsible for 16% of lifetime bladder cancers among the exposed population, while Regli et al. (2015) calculated a value of 14%. The TTHM exposure concentration assumed by the two analyses (38.5 μg/L) is close to the 35.4 μg/L average concentration in drinking water from the most recent EPA national drinking water quality survey (USEPA, 2016).

Health Risk Assessment: Impact on Article Conclusions

Health risk information was used by Good and VanBriesen (2019) to determine how far downstream to carry out transport modeling. The Regli et al. (2015) calculations of health effect levels are consistent with standard practice. The conclusion of Regli et al. (2015) that an increase of TTHM on the order of 1 µg/L is associated with an excess bladder cancer risk of 10^{-4} is of concern, given that most drinking water systems exceed this concentration.

Table 8-1
Comparison of EPA (2005) and Regli et al. (2015) evaluations of TTHM association with bladder cancer risk

	EPA (2005)	Regli et al (2015)	Comparable?
Source of risk values	Meta-analysis of data from five individual studies (Villanueva et al., 2003)	Pooled analysis of data from six individual studies (Villanueva et al., 2004)	No - different approaches
Study locations	US (1), Canada (1) France (1), Finland (1)	U.S. (2), Canada (1), France (1), Finland (1), Italy (1)	Yes
Publication years of studies	1987 – 1998	1993 - 1998, 2003 unpublished data	Yes
Exposure metric	Years of exposure (ever exposed to chlorinated surface water)	TTHM concentration. Derivation varied among studies. Some extrapolated current TTHM levels to past exposure. Others estimated levels based on a DBP formation model or from mutagenicity data.	No - different exposure metrics
Assumed exposure period	40 years before Stage 1 DBP rulemaking	40 years prior to Villanueva interviews with cases and controls	Yes, but Villanueva (2004) assumed more recent exposure
TTHM exposure level for EPA and EPRI PAR calculations	EPA assumed 38.05 µg/L (pre-Stage 1 average)	Averages in studies that measured TTHM ranged from 0.6 to 32.2 µg/L. Regli et al. (2015) used 38.05 µg/L in the PAR calculation.	Same value used in PAR calculation, likely overestimates current exposure.
Population attributable risk (PAR)[1]	15.7%	14%	Yes

[1] *Population attributable risk* (PAR) is the proportion of the incidence of a disease in the *population* (exposed and unexposed) that is due to exposure. It is the incidence of a disease in the *population* that would be eliminated if exposure were eliminated.

9
BROMIDE CONCENTRATIONS OF CONCERN

Good and VanBriesen (2019) applied an additional factor of 10 to the 10 µg/L bromide concentration identified by Regli et al. (2015) as the lowest concentration with an observable increase in TTHM. Bromide transport downstream was modeled until the total concentration contributed by all upstream power plants fell below 1 µg/L at a DWTS raw water intake. The stated rationale for the factor of 10 was to provide additional conservatism to offset use of median river flow rates. There are several concerns with this approach:

- The application of a safety factor to account for flow rate variations should not be necessary when the endpoint of concern is cancer, a chronic health effect. Drinking water consumers are not exposed to low flow conditions over a lifetime; a median flow rate is more representative of chronic exposures.
- Bromide contributions were tracked up to hundreds of miles downstream, which creates increased uncertainty due to the complexity of hydraulics/hydrology of riverine systems that affects dilution.
- The lowest method detection limits of commonly used methods for measuring bromide are in the range of 10 to 20 µg/L. Quantitation limits (concentrations at which accurate results can be obtained) are typically much higher. Thus, it is not possible with conventional analytical methods to measure bromide accurately at 1 µg/L, preventing any useful validation of the modeling results.
- The concentrations identified as contributing to potential risks are below background levels of bromide in many areas of the country. Background bromide measurements are available from water quality monitoring stations that are part of the National Network of Reference Watersheds (USGS, 2018). These watersheds are minimally disturbed by anthropogenic influences; however, there could be unidentified bromide sources contributing to the observed concentrations. The concentrations in these watersheds ranged from below detection limits up to 1,700 µg/L, even in locations with no potential influence from power plant effluent. This analysis was based on 904 bromide samples collected at reference watershed locations in 12 states. These samples represent all the bromide measurements in the reference watershed database as of May 8th, 2019. Average bromide concentrations reported for these reference watersheds exceed 20 µg/L in nine of the 12 states (Figure 9-1). Pennsylvania has the most reference watershed bromide data (591 samples collected); concentrations ranged from 20 µg/L to 100 µg/L. The value for Kansas (1,700 µg/L) is truncated in the figure to make the lower values more visible. While the reference watersheds are considered minimally disturbed, they may be potentially influenced by activities such as hydraulic fracturing. Nonetheless, these background levels call into question the relevance of tracking FGD wastewater bromide contributions down to 1 µg/L.
- An evaluation of EPA's Fourth Unregulated Contaminant Monitoring Rule (UCMR 4) bromide data (USEPA, 2019b) suggests that a large proportion of the bromide data collected nationwide exceeds detection limits, and thus also exceeds the 1 µg/L threshold used by Good and VanBriesen (2019). Figure 9-2 provides a frequency distribution of the most recent UCMR 4 data (as posted on the US EPA website as of January 2019). The median

concentration is 22 µg/L, barely above the 20 µg/L method detection limit. Bromide concentrations exceed 50 µg/L in 24% of these samples. If these levels are indeed of concern, these data suggest that the bromide problems extend far beyond FGD wastewater bromide contributions.

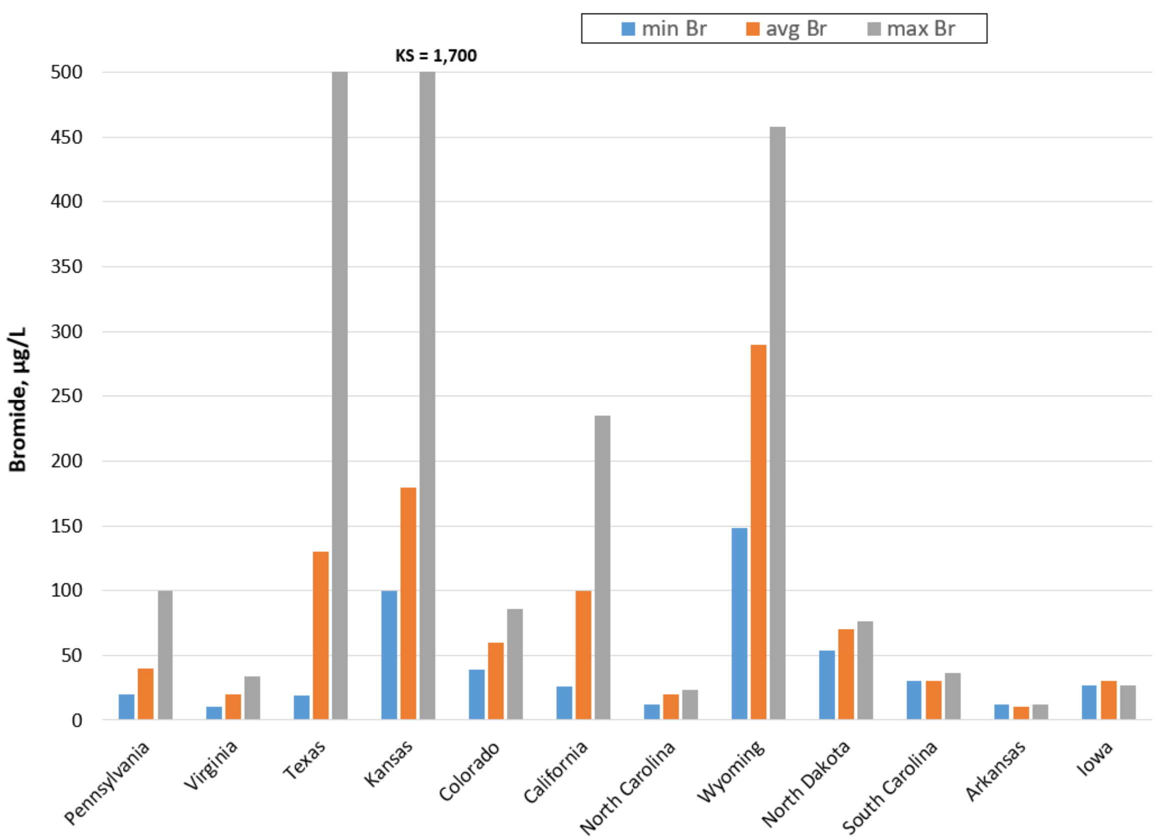

Figure 9-1
Bromide concentrations in reference watersheds

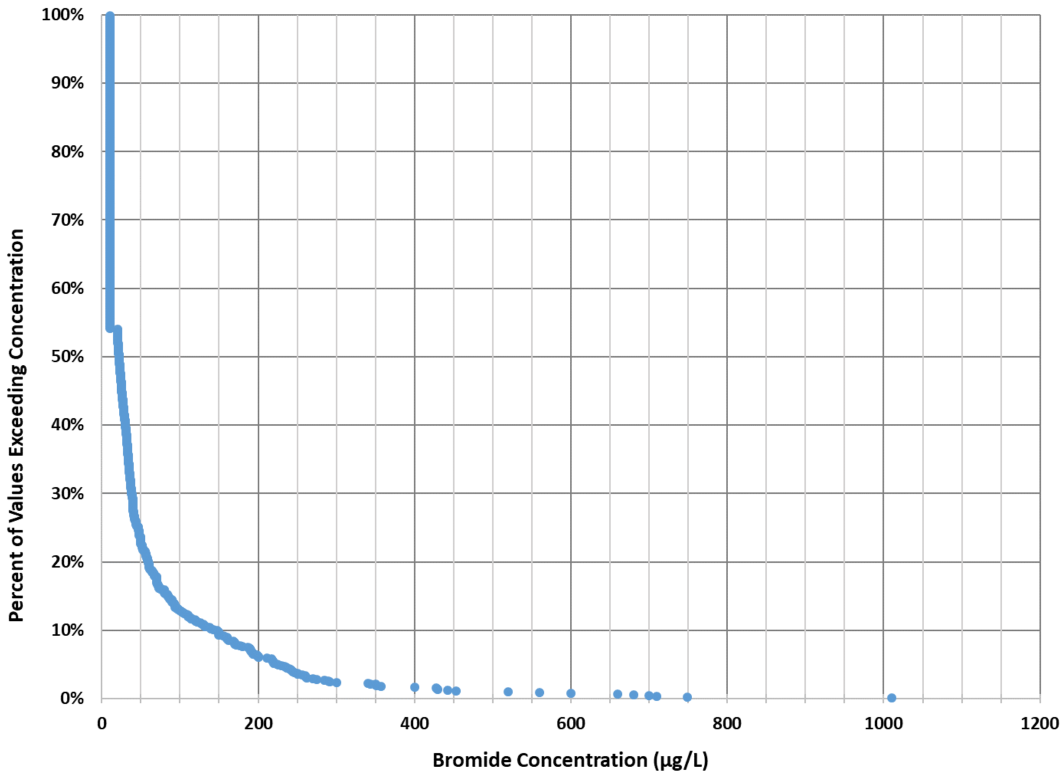

Figure 9-2
Cumulative distribution plot of bromide in UCMR 4 samples (through January 2019)

Bromide Concentrations of Concern: Impact on Article Conclusions

The application by Good and VanBriesen (2019) of a 10x safety factor resulted in a 1 µg/L bromide contribution threshold for downstream modeling. This is problematic for several reasons:

- This threshold resulted in some extremely long flow paths, creating potential inaccuracy of transport modeling.
- Such low concentrations are not measurable by standard analytical techniques, preventing the validation of the model results.
- Background levels in many areas of the country, including those considered to be reference watersheds, may be as high as or higher than the proposed levels of concern.

10
POTENTIALLY AFFECTED POPULATIONS

Good and VanBriesen (2019) evaluated potential vulnerability of downstream drinking water users to effects from upstream FGD bromide contributions. This assessment used a non-standard vulnerability metric ("cumulative wet FGD Br effect") calculated by multiplying the predicted FGD bromide concentration contributions from all upstream power plants by the potentially affected population of downstream drinking water users. The population vulnerability metric used by Good and VanBriesen (2019) is difficult to interpret and has little practical meaning (the units correspond to "population exposed - µg/L").

To calculate this metric, Good and VanBriesen (2019) summed the modeled bromide contributions from all upstream power plants contributing ≥ 1 µg/L Br at each HUC 12 watershed within a HUC 4 subregion. The total bromide contribution was then multiplied by the population served by the DWTS within that HUC 12 watershed to obtain a population effect value. The HUC 12 effect values were then summed to obtain the effect value for the entire HUC 4 subregion. This is problematic for several reasons.

First, there is much uncertainty in the quantification of affected downstream populations, as it is difficult to determine from which exact stream segment a drinking water plant withdraws water. For instance, an intake in a HUC 12 downstream from an FGD discharge may not necessarily be located on a stream segment that is on a downstream flow path from that FGD. Secondly, the percentage of a population served by the surface water intake of interest rather than from a different source used by the same drinking water plant (e.g., groundwater, alternate surface water) can be difficult to determine. Thirdly, the map of impacted subregions (Figure 4 in Good and VanBriesen, 2019) indicated a higher potential population risk than the modeling predicts, due to the way at-risk watersheds were aggregated to the HUC 4 subregion level. For example, Figure 10-1 shows a HUC 4 subregion in Missouri that is marked as entirely impacted in Good and VanBriesen (2019), Figure 4. However, only six of the 293 HUC 12 watersheds in that subregion (highlighted in yellow) were flagged by the model results as affected by FGD bromide contributions. Lastly, the level of bromide in the DWTS source water does not have a direct relationship with the DBP exposure at the customer location. Numerous factors come into play both within the drinking water plant and in the distribution system that can impact DBP concentrations, and hence exposures, at the tap. Thus, relating the bromide concentration to population impact is not a valid predictor of health risk.

Population Effect: Impact on Article Conclusions

The procedure used by Good and VanBriesen (2019) to estimate the potentially affected populations involves a high degree of uncertainty and likely overestimates the potential impacts and risk to downstream water users due to several factors.

- The population vulnerability metric is difficult to interpret and has little practical meaning (the units correspond to "population exposed - µg/L").

- A DWTS intake may be in a HUC 12 downstream from an FGD discharge, but may not necessarily be on the downstream flow path from that discharge.
- Some percentage of the population served by a DWTS may be served by a source other than the primary surface water intake (e.g., groundwater, alternate surface water).
- The map of impacted subregions (Figure 4 in Good and VanBriesen, 2019) indicated a higher potential population risk than the modeling predicts, due to the way at-risk watersheds were aggregated to the HUC 4 subregion level.
- The level of bromide in the DWTS source water does not have a direct relationship with the DBP exposure at the customer location.

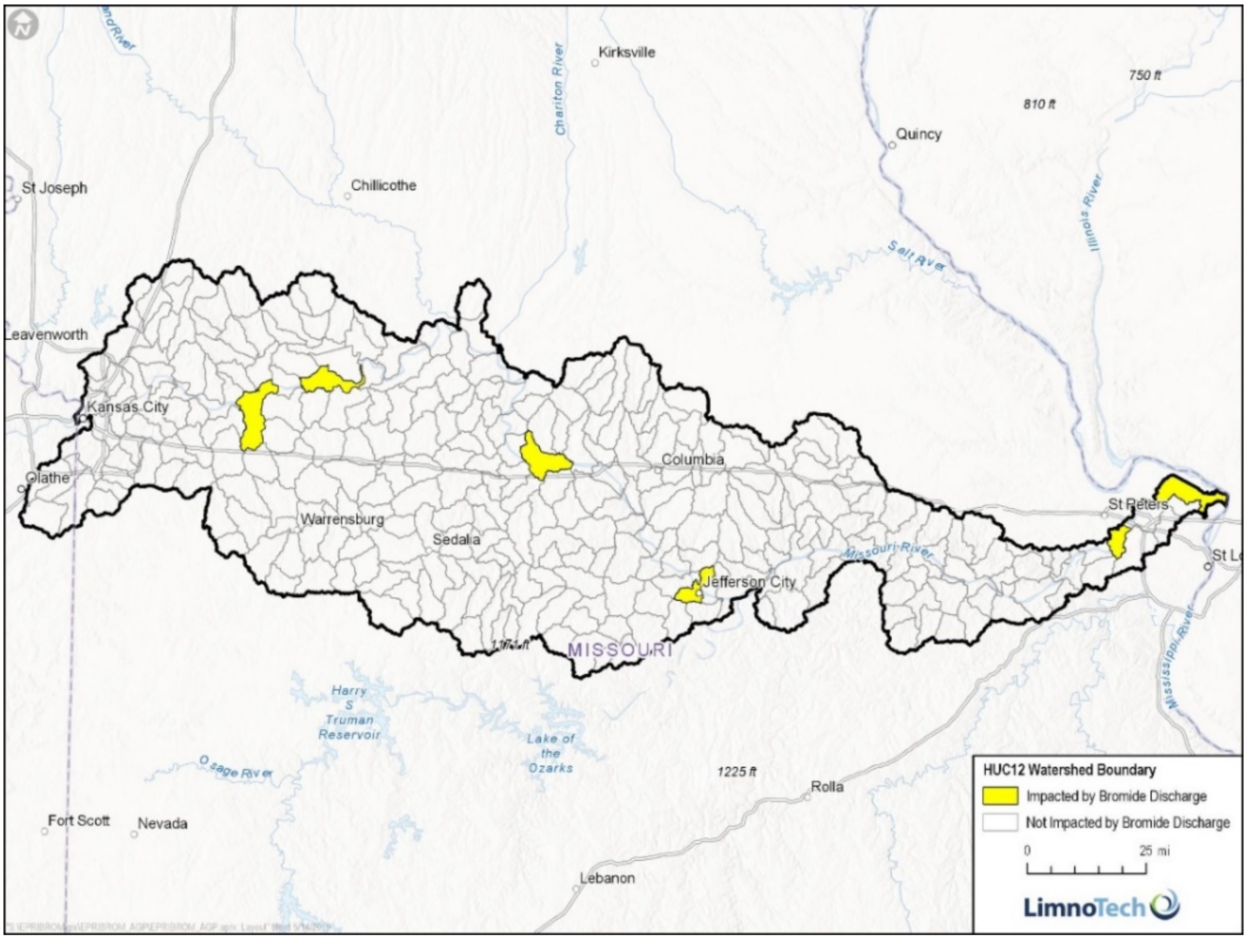

Figure 10-1
Example of a HUC 4 subregion labeled as impacted in Good and VanBriesen (2019)

11 CONCLUSIONS

EPRI reviewed several articles which modeled the discharge of bromide from power plant flue gas desulfurization (FGD) wastewater discharges, transport to downstream drinking water treatment intakes, and potential human health risks due to formation of disinfection byproducts. Modeling the source term and transport of substances for even a single well-characterized source and surface water body is difficult; modeling at the watershed or national level requires many simplifying assumptions. The use of conservative assumptions to compensate for uncertainty is a common approach in health risk assessments, where the intent is to be protective of human health, but can lead to unrealistic outcomes when multiple conservative assumptions are combined. EPRI's review is intended to improve the accuracy of such Br discharge modeling and to inform future modeling efforts.

The authors used information available at the time of publication to estimate downstream impacts for a large number of U.S. coal-fired power plants. Due to more recent retirements of coal-fired units, changes in power industry practices, and the use of conservative simplifying assumptions in the models, the articles overestimate the regional or national impact of power plant bromide discharges.

- One-quarter of the power plants modeled by Good and VanBriesen (2019) and three of seven plants on the Ohio River modeled by Cornwell et al. (2018) are no longer in operation or do not discharge FGD wastewater to surface water.
- Based on surveys of power companies, the amount of bromide added to refined coal or for mercury control in recent years by plants that discharge to surface water is lower than assumed by Good and VanBriesen (2019) and far lower than assumed by Cornwell at al. (2018).
- On a nationwide basis, the native concentrations of bromide in coal are lower than the 75th percentile concentrations used by Good and VanBriesen (2019) to model downstream population vulnerability and much lower than assumed by Cornwell at al. (2018).
- Good and VanBriesen (2016; 2019; and 2017 by reference) contain an equation with an error in calculating coal tonnage which, if used in modeling bromide mass loadings to surface water, would overestimate the mass of bromide by 14% for bituminous coal, 88% for subbituminous coal, 130% for lignite coal, and 42% for refined coal. Cornwell et al. (2018) cites the Good and VanBriesen (2016) methodology; thus, it is possible that Cornwell et al. (2018) also contains the error.
- Good and VanBriesen (2019) procedures for estimating bromide concentrations in native and refined coal based on coal rank overestimate mass loadings to surface water by 5 to 58% for the 75th percentile modeling scenario. These overestimates are in addition to the overestimates associated with the error in the coal tonnage calculation.
- The hydrologic modeling performed by Good and VanBriesen (2019) used a simplistic modeling approach that may have underestimated dilution. Cornwell et al. (2018) focused on low flow conditions, which may be appropriate for modeling potential short-term

exceedances of drinking water DBP limits, but not for evaluating long-term health risk. Neither model was validated with in-stream monitoring data.

- The assumption by Good and VanBriesen (2019) that the bromide concentration at the intake of a drinking water treatment plant is proportional to population risk is not valid. Many factors within the treatment plant and in the water distribution system influence the levels of DBPs reaching the consumer. Among these are factors that promote or inhibit DBP formation, such as water temperature, pH, and the concentration of natural organic matter; as well as treatment processes such as aeration that remove some DBPs. In addition, as discussed by Regli et al. (2015), there is not a fixed relationship between bromide concentration and DBP formation; as the bromide concentration increases, proportionally less DBP may be formed due to limited natural organic matter precursors.
- The total bromide concentrations contributed by upstream power plant sources that are considered significant by Regli et al. (2015) and Good and VanBriesen (2019) are within the range of concentrations observed in relatively uncontaminated (reference) water bodies. If these concentrations are indeed of concern, monitoring data suggest that sources of bromide that contribute to that concern are not limited to the power industry, including not only other anthropogenic bromide sources such as oil recovery and road salt, but also natural geologic formations.

The most accurate approach to determine impacts of upstream bromide sources on a downstream drinking water facility is to use a watershed approach using monitoring data from each source, site-specific hydrologic data, and measurement of bromide and DBPs in downstream drinking water system influent and water delivery systems. However, that is an extremely complex and expensive approach that must be carried out with a high frequency over multiple seasons to reflect source and flow variations. There is a role for modeling to screen for potential problems on a regional or national basis; but the modeling approach should avoid using multiple conservative assumptions to avoid unrealistic estimates of downstream impact. It would greatly improve the confidence in the models to conduct several site-specific studies to validate the model predictions with observed data. It is hoped that the observations in this report will assist in improving future modeling efforts by identifying areas where more recent data (e.g., bromide addition rates), a more detailed analysis of existing data (e.g., coal subrank shipments), or a more sophisticated approach (hydrologic modeling) would improve the analysis.

12
REFERENCES

Buschmann, J., L. Lindau, and B.W. Vosteen. 2005. "The KNX Coal Additive Technology, A Simple Solution for Mercury Emissions Control." ALSTOM. https://www.vosteen-consulting.de/fileadmin/user_upload/Publikationen/2005/2005_12_Vosteen_Consulting_KNX_TM_Coal_Additive_Technology_PowerGen_USA.pdf

Chang, R.; Dombrowski, K.; & Senior, C., 2010. Near and Long-Term Options for Controlling Mercury Emissions from Power Plants. URS Corporation, San Francisco.

Cornwell, D. A., Baljit, K. S., Brown, R., and McTigue, N. E. 2018. Modeling Bromide River Transport and Bromide Impacts on Disinfection Byproducts, *Journal of the American Water Works Association*, Volume 110, Issue 11, November.

EPRI. 2009. Mercury Control Update. EPRI. Palo Alto, CA: 2009. 1017564.

EPRI. 2014a. *Program on Technology Innovation: Bromine Usage, Fate, and Potential Impacts for Fossil Fuel–Fired Power Plants.* EPRI, Palo Alto, CA: 2014. 3002002510.

EPRI. 2014b. *2014 Update on EPRI's Balance-of-Plant Effects Study of Bromine-Based Mercury Controls.* EPRI, Palo Alto, CA: 2014. 3002003404.

EPRI. 2015. *Halogen Addition for Mercury Control and Related Balance-of-Plant Issues*. EPRI, Palo Alto, CA: 2015. 3002006157.

EPRI. 2016. *Halogen Addition for Hg Control and Related BOP Issues: 2016*. EPRI, Palo Alto, CA: 2016. 3002008332.

EPRI. 2017. *Mercury and Air Toxics Control: 2017 Update*. EPRI, Palo Alto, CA: 2017. 3002010378.

Good, K. D., and VanBriesen, J. M. 2016. Current and Potential Future Bromide Loads from Coal-Fired Power Plants in the Allegheny River Basin and Their Effects on Downstream Concentrations. *Environ. Sci. Technol.* 2016, 50, 9078−9088.

Good, K. D., and VanBriesen, J. M. 2017. Power Plant Bromide Discharges and Downstream Drinking Water Systems in Pennsylvania. *Environ. Sci. Technol*. 2017, 51, 11829−11838.

Good, K. D., and VanBriesen, J. M. 2019. Coal-Fired Power Plant Wet Flue Gas Desulfurization Bromide Discharges to U.S. Watersheds and Their Contributions to Drinking Water Sources, *Environ. Sci. Technol*., 53, 213-223, and Supporting Information.

Kolker, A., J. C. Quick, C.L. Senior, and H.E. Belkin. 2012. *Mercury and Halogens in Coal—Their Role in Determining Mercury Emissions from Coal Combustion*. Fact Sheet 2012–3122. U.S. Geological Survey.

Kolker, A. and Quick, J. 2015. *Mercury and Halogens in Coal. Mercury Control for Coal-Derived Gas Streams* (Granite, E.; Pennline, H.; & Senior, C., editors). Wiley-VCH Verlag GmbH & Co., Weinheim, Germany.

Meij, R. 1994. Trace element behavior in coal-fired power plants. *Fuel Processing Technology.* Volume 39, Issues 1–3, August 1994, Pages 199-217.

Ohio River Valley Water Sanitation Commission (ORSANCO). 2014. Characterization of dissolved solids in the Ohio River and selected tributaries. Cincinnati, OH.

Ohio River Valley Water Sanitation Commission (ORSANCO). 2019. Bimonthly Data: 2000 – Present. www.orsanco.org/data/bromide. Accessed June 21, 2019.

Quick, J. 2019. Personal communication with Jeffery Quick, Utah Geologic Survey. Shipping totals to EGUs by subrank calculated from EIA 2017 Form 923 data and heating rates, assigned by county following methodology described in: Quick, J. C. and E. Marland. Systematic error and uncertain carbon dioxide emissions from U.S. power plants., Supporting Information, *J. Air & Waste Management Association*. Volume 69, 2019 - Issue 5.

Regli, S. and 8 co-authors. 2015. Estimating Potential Increased Bladder Cancer Risk Due to Increased Bromide Concentrations in Sources of Disinfected Drinking Waters. *Environ. Sci. Technol*., 49, 13094−13102, and Supporting Information.

Srivastava, R. K., N. Hutson, B. Martin, F. Princiotta, J. Staudt. 2006. Control of mercury emissions from coal-fired electric utility boilers. *Environ. Sci. Technol.* 2006, 40 (5), 1385−1393.

U.S. Environmental Protection Agency (USEPA). 2005. Appendices to the Economic Analysis for the Final Stage 2 Disinfectants and Disinfection Byproducts Rule Volume I (A-E2).

U.S. Environmental Protection Agency (USEPA). 2016. Six-Year Review 3 Compliance Monitoring Data (2006-2011). TTHM data from THM.zip file, substituting 10 µg/L (typical detection limit) for nondetect data. https://www.epa.gov/dwsixyearreview/six-year-review-3-compliance-monitoring-data-2006-2011

U.S. Environmental Protection Agency (USEPA). 2019a. Integrated Risk Information System (IRIS). Accessed August 1, 2019.

U.S. Environmental Protection Agency (USEPA). 2019b. Fourth Unregulated Contaminant Monitoring Rule (2018 – 2020) Occurrence Data. Accessed May 15, 2019. https://www.epa.gov/dwucmr/occurrence-data-unregulated-contaminant-monitoring-rule#4

U.S. Geological Survey (USGS). 2018. National Network of Reference Watersheds. Accessed May 8, 2019. https://my.usgs.gov/nnrw/main/home

VanBriesen, J.M. 2019. Personal communication to Jeff Thomas, EPRI. October 26, 2019
Villanueva, C.M., F. Fernandez, J.O. Grimalt, M. Kogevinas. 2003. Meta-analysis of studies on individual consumption of chlorinated drinking water and bladder cancer**.** *J Epidemiol Community Health.* 2003**.** 57:166–173**.**

Villanueva, C.M., K.P. Cantor, S. Cordier, J.J. Jaakkola, W.D. King, C.F. Lynch, S. Porru, M. Kogevinas. 2004. Disinfection Byproducts and Bladder Cancer, A Pooled Analysis. *Epidemiology.* Volume 15, Number 3, May 2004.

Wood, G.H. et al., *Coal Resource Classification System of the U.S. Geological Survey*, Geological Survey Circular 891, Second Printing, 1992.

Export Control Restrictions

Access to and use of this EPRI product is granted with the specific understanding and requirement that responsibility for ensuring full compliance with all applicable U.S. and foreign export laws and regulations is being undertaken by you and your company. This includes an obligation to ensure that any individual receiving access hereunder who is not a U.S. citizen or U.S. permanent resident is permitted access under applicable U.S. and foreign export laws and regulations.

In the event you are uncertain whether you or your company may lawfully obtain access to this EPRI product, you acknowledge that it is your obligation to consult with your company's legal counsel to determine whether this access is lawful. Although EPRI may make available on a case by case basis an informal assessment of the applicable U.S. export classification for specific EPRI products, you and your company acknowledge that this assessment is solely for informational purposes and not for reliance purposes.

Your obligations regarding U.S. export control requirements apply during and after you and your company's engagement with EPRI. To be clear, the obligations continue after your retirement or other departure from your company, and include any knowledge retained after gaining access to EPRI products.

You and your company understand and acknowledge your obligations to make a prompt report to EPRI and the appropriate authorities regarding any access to or use of this EPRI product hereunder that may be in violation of applicable U.S. or foreign export laws or regulations.

The Electric Power Research Institute, Inc. (EPRI, www.epri.com) conducts research and development relating to the generation, delivery and use of electricity for the benefit of the public. An independent, nonprofit organization, EPRI brings together its scientists and engineers as well as experts from academia and industry to help address challenges in electricity, including reliability, efficiency, affordability, health, safety and the environment. EPRI also provides technology, policy and economic analyses to drive long-range research and development planning, and supports research in emerging technologies. EPRI members represent 90% of the electricity generated and delivered in the United States with international participation extending to 40 countries. EPRI's principal offices and laboratories are located in Palo Alto, Calif.; Charlotte, N.C.; Knoxville, Tenn.; Dallas, Texas; Lenox, Mass.; and Washington, D.C.

Together…Shaping the Future of Electricity

3002017477