

Anthropogenic and Geogenic Contaminant Bioexposures Affecting Aquatic Ecosystems

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Chapter C of

Knowledge Gaps and Opportunities in Water-Quality Drivers of Aquatic Ecosystem Health

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Supplemental Information

Concentrations of chemical constituents in water are given in either milligrams per liter (mg/L) or micrograms per liter ($\mu\text{g/L}$).

Abbreviations

ACCWW	USGS accumulated wastewater ratio in-line mapper
AGC	anthropogenic and geogenic contaminant
ALB	aquatic life benchmarks
BAC	bioaccumulation coefficient
BCF	bioconcentration factor
BLM	biotic ligand model
CCC	criterion continuous concentration
CEC	contaminant of emerging concern
CMC	criterion maximum concentration
CWA	Clean Water Act
DART	decision analysis by ranking techniques
DOC	dissolved organic carbon
DOM	dissolved organic matter
EC	efficiency concentration
EPA	U.S. Environmental Protection Agency
EPI	EPA Estimation Parameter Interface Suite
FA	fulvic acid
FIAM	free ion activity model
GIS	geographic information system
HA	humic acid
IWS	USGS Integrated Water Science
K_d	distribution coefficient
K_{ow}	octanol-water coefficient
LC	lethal concentration
LOEC	lowest observed effect concentration
MCL	maximum contaminant level
MeHg	methylmercury

NGWOS	USGS Next Generation Water Observing Stations
NOEC	no observed effect concentration
NWQP	USGS National Water Quality Program
PBTO	persistence, bioaccumulation, toxicity, and environmental occurrence
PFAS	per- and polyfluoroalkyl substances
PNEC	predicted no effect concentration
QSAR	quantitative structure-activity relationship
TMDL	total maximum daily load
USGS	U.S. Geological Survey
WMA	USGS Water Resources Mission Area
WWTP	wastewater treatment plant
XAS	X-ray adsorption spectrometry

Chemical Symbols

Sb	antimony
As	arsenic
Bi	bismuth
Cd	cadmium
C	carbon
Cr	chromium
Co	cobalt
Cu	copper
Fe	iron
Pb	lead
Li	lithium
Mn	manganese
Hg	mercury
Mo	molybdenum
O	oxygen
Se	selenium
Ti	titanium
Zn	zinc

Chapter C

Anthropogenic and Geogenic Contaminant Bioexposures Affecting Aquatic Ecosystems

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Purpose and Scope

This chapter addresses knowledge gaps that, if filled, could improve predictions of aquatic ecosystem health as affected by anthropogenic and geogenic contaminant bioexposures. The gaps identified in this chapter are not intended to be comprehensive but are instead focused on key opportunities to fill those gaps for the U.S. Geological Survey (USGS) Water Resources Mission Area (WMA, <https://www.usgs.gov/mission-areas/water-resources>). Anthropogenic and geogenic contaminant effects on beneficial uses of water are not addressed in this chapter but are covered in Chapters C and D of Tesoriero and others (2024), the companion Open-File Report to this publication.

Statement of the Problem

Aquatic organisms are exposed to anthropogenic and geogenic contaminants (AGC) in water bodies via direct contact and (or) through diet, a process globally defined here as bioexposure. AGCs negatively affect aquatic ecosystems, including species diversity, and resiliency to water hazards (Chapman and others, 2010; Brack and others, 2019; Zhang and others, 2023). The economic effects of AGCs are significant due to the costs of restoration and mitigation of U.S. Environmental Protection Agency (EPA) Superfund sites, necessary infrastructure upgrades to wastewater and drinking water facilities, and loss of recreational and commercial fisheries (EPA, 2016, 2017, 2021a, 2021b). Furthermore, there are undesired consequences from water management projects (for example, water diversions) and habitat restoration where concentrations of AGCs may be altered, causing greater ecosystem harm (for example, wetland restoration enhancing mercury [Hg] methylation) (Chapman and others, 2010).

Water bodies where concentrations of some AGCs exceed established state water quality standards are listed under section 303(d) of the Clean Water Act (CWA; 33 U.S.C 1251 et seq.; <https://www.epa.gov/tmdl>), whereby States are required to rank and prioritize those waterbodies for the development of total maximum daily loads (TMDL). Following nutrients and sediments, AGCs such as Hg are listed as a primary cause of a water body being listed under the CWA. Yet, there are challenges

associated with the CWA listing and the TMDL process. Federal and State water quality standards may not exist for AGCs detected in the water body, and for those that do have existing standards they may not be strict enough to protect aquatic life and resources. Quantifying AGCs in the environment is only the first step in establishing the significance of the detected AGCs and how they affect aquatic ecosystem health. Studying and understanding uptake mechanisms of AGCs is key to predicting contaminant fate and minimizing toxic effects. Reducing and mitigating AGC in water bodies is complex and requires detailed understanding of physical, chemical, and biological processes controlling the fate and effects of AGC loads entering ecosystems.

Regulators and resource managers require understanding and tools for: (1) attributing AGC loads to their sources, (2) quantifying AGC concentrations in space and time, (3) quantifying and incorporating direct and indirect factors (for example, biotic, abiotic) controlling bioexposure into coupled hydrologic-biogeochemical models, (4) linking bioexposures and effects, and (5) predicting changes in AGC fate in response to changing human and natural influences. Addressing these needs requires an approach that integrates investigation of physical and biogeochemical processes with data collection and model development.

While there are many key limitations in our understanding of mechanisms of anthropogenic and geogenic bioexposures and how they affect human and aquatic ecosystem health, the ecosystem health gap analysis team for the WMA Water Quality Program identified the following four major gaps grouped by anthropogenic or geogenic contaminant class (table C1):

- Anthropogenic contaminants of emerging concern (CECs)
 1. Major factors affecting human and aquatic organism health, and
 2. Water quality assessment tools for aquatic organism and human health risk.
- Geogenic contaminants
 1. Mechanistic understanding of uptake dynamics of geogenic contaminants for coupled hydrologic-biogeochemical watershed assessments and model development, and

2 Water-Quality Processes Affecting Aquatic Ecosystem Health

Table C1. The four major bioexposure knowledge gaps of water quality drivers of aquatic ecosystem health identified by the U.S. Geological Survey (USGS), grouped by anthropogenic contaminants of emerging concern (CECs) or geogenic contaminant class.

[WWTP, wastewater treatment plant; LC50, 50 percent lethality concentration; BAC, bioaccumulation coefficients; BCF, bioconcentration factors; QSAR, quantitative structure-activity relationship; EPA, U.S. Environmental Protection Agency; K_d , distribution coefficient; DOM, dissolved organic matter; IWS; Integrated Water Science.]

What	Gap	Why	How	References
Anthropogenic Contaminants				
Major factors affecting human and aquatic organism health	Anthropogenic water use, specifically wastewater treatment plants (WWTP), which create complex mixtures of contaminants with unknown ecotoxicological effects.	WWTP discharge and toxicity benchmark data (that is, LC50, BAC, BCF) need to be consolidated in an accessible way to develop biogeochemical model parameters and improve understanding of bioexposure effects of complex mixtures of anthropogenic contaminants.	Laboratory-based aquatic organism feeding experiments in complex mixtures such as those in WWTP waters, along with in-situ mobile laboratory experiments to improve tools such as QSAR models.	Kolpin and others, 2002; Barnes and others, 2008; Focazio and others, 2008; Vajda and others, 2011; Bradley and others, 2016; Croteau and others, 2016.
Water quality assessment tools for aquatic organism and human health risk	Understanding of anthropogenic contaminant bioexposure and uptake is lacking and major improvements to databases and water quality tools are needed.	Predictive capabilities of anthropogenic contaminant bioavailability are limited. An ever-expanding list of contaminants and the complex biogeochemical processes at play need to be better accounted for in widely used models.	Compilation of an expanded database of contaminants of concern, in combination with laboratory and field studies to improve prediction capabilities. Development of screening tools including proxy measurements, computational toxicology tools and predictive modelling.	EPA, 2012, 2016, 2021a, 2021b; Faunce and others, 2023; Barber and others, 2022; Pavan and Worth, 2008a & 2008b; Ortiz de García and others, 2013; Omar and others, 2016; Fischer and others, 2017; Brack and others, 2019.
Geogenic Contaminants				
Mechanistic understanding of uptake dynamics of geogenic contaminants for coupled hydrologic-biogeochemical watershed assessments and model development	Methods to enhance understanding of factors and processes controlling uptake and bioavailability of geogenic contaminants for watershed assessments of risk and predictive modeling.	Partitioning of geogenics based on traditional assessments of total concentrations in water and particulate matter (that is, K_d) do not adequately assess risk at levels required by the regulatory community or numerically predict bioaccumulation in nature across a range of environmental conditions.	Field analysis of chemical speciation (that is, redox state, complexes), preservation, particle characteristics and hydrologic residence time along hydrologic-geochemical gradients coupled with chemical speciation modeling. Verify process controls on biouptake by laboratory feeding studies.	Luoma and Rainbow, 2009; Mebane and others, 2020; Ponton and others, 2020; Stewart and others, 1999; Croteau and others, 2005b & 2016. Nowell and others, 2014; Sangion and Gramatica, 2016.
Bioavailability and toxicity of organometallic complexes	Assess the bioavailability toxicity of metal-dissolved organic matter complexes, on a basin-by-basin case and incorporate results into more robust geochemical and ecotoxicological models.	Current models only account for total metal concentrations and the fraction that binds to biotic surfaces. Studies highlight that these complexes may be more bioavailable than believed and our understanding of metal uptake and toxicity needs updating.	A variety of in-situ and laboratory-based experiments to measure metal-DOM complex uptake and toxicity at the base of the food web (that is, microbes, algae). Experiments will vary with respect to metal, DOM source and biota based on IWS basin site specific needs.	Lamelas and others, 2005; Wang and others, 2019; Mebane and others, 2020; Stewart and Malley, 1999; Roditi and others, 2000; Sánchez-Marín and others, 2007; Parkhurst and Appelo, 2013.

- Bioavailability and toxicity of organometallic complexes.

Status of Knowledge and Key Limitations

Broad factors that contribute to the bioexposure of AGC in aquatic ecosystems include sources and drivers, and mechanistic and biogeochemical processes (fig. C1). The introduction of AGC into aquatic ecosystems is influenced by contaminant sources (that is, urban land use, industrial, mining, and agricultural processes, and so forth) and drivers, such as extreme events related to climate change (that is, wildfires, droughts, and so forth) and land management practices (that is, wetland restoration, dams, irrigation, and so forth) (Kolpin and others, 2002; Barnes and others, 2008; Focazio and others, 2008; Bradley and others, 2016). These and other biogeochemical processes (that is, sorption, precipitation, speciation, redox changes, and so forth) determine loads, timing, and magnitude of concentrations of AGCs.

Within the receiving environment, ecosystem effects of AGCs are in part determined by mechanistic processes that influence contaminant transfer through the aquatic food web (fig. C1). Uptake from the dissolved phase into the base of the food web (that is, microorganisms, algae, detritus, sediment,

and so forth) is influenced by (1) site-specific water quality (that is, temperature, dissolved oxygen [O], conductivity, oxidation-reduction potential, and turbidity), (2) concentration of the contaminant in the water column, and (3) physical and chemical properties of the contaminant that affect AGC transport, distribution and residence time within the hydrologic system, and (for metals) speciation. The capacity of a constituent to be taken up by a living organism and integrated into metabolic processes, or the fraction of the concentration that is absorbed and (or) adsorbed determines the “bioavailability” of a contaminant (de Paiva Magalhães and others, 2015). Certain AGCs that are consumed or absorbed by organisms that cannot catabolize or excrete quickly enough results in a buildup of the toxic chemical inside the organism, known as bioaccumulation. When the increasing concentration of a toxic chemical is transferred up the food chain, this process is known as biomagnification.

Contaminant transfer from the base of the food web occurs via the diet with trophic pathways playing an important role in determining bioexposures. Where an organism eats and diet composition can significantly alter bioexposures (Stewart and others, 2004), which highlights the important roles of ecology (Johnson and others, 2020), organism biology (Wang and others, 1996; Schlekot and others, 2002), and species invasions (Eagles-Smith and others, 2008; Lepak and others, 2009; Lepak and others, 2015) in providing an integrated assessment of contaminant

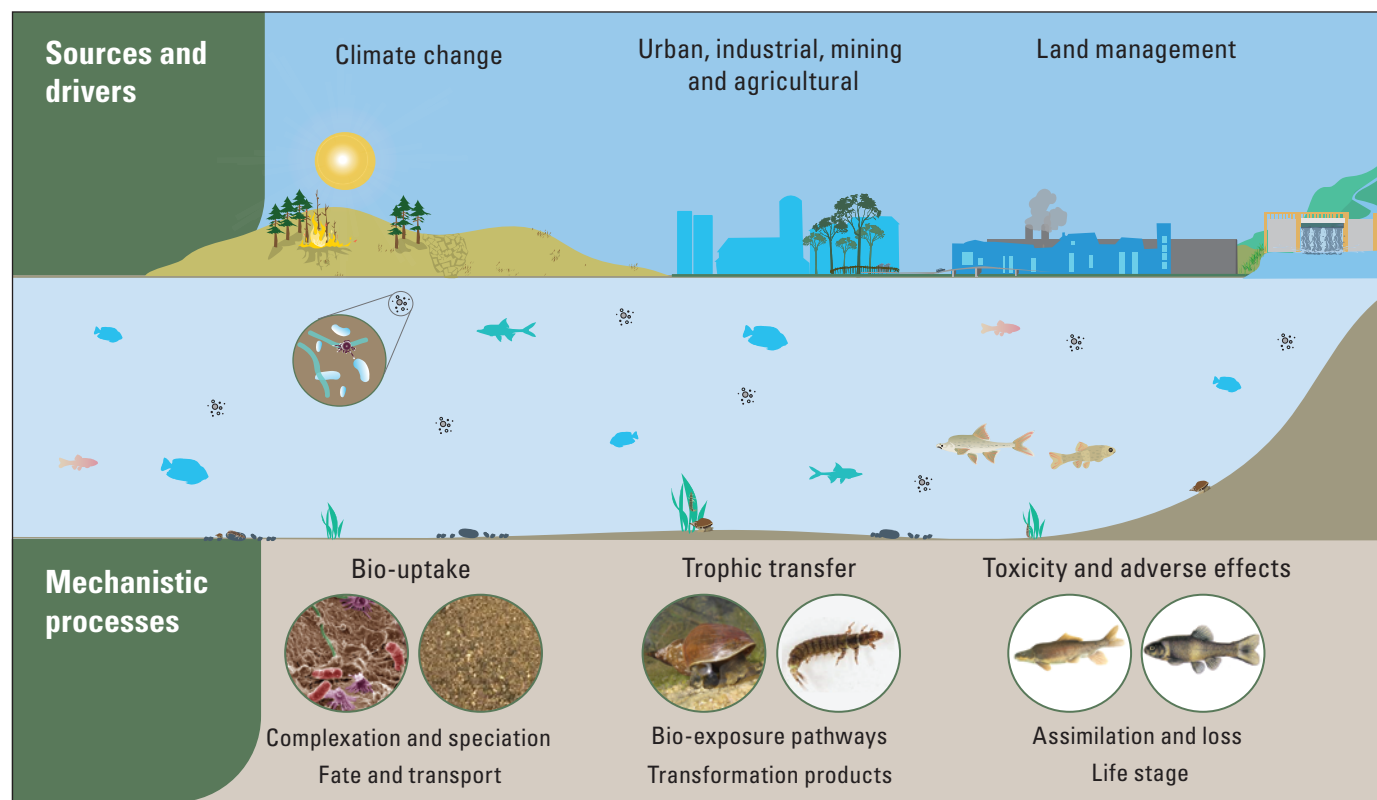


Figure C1. Drivers and mechanisms of bioexposure of contaminants in aqueous ecosystems illustrating different ways contaminants enter the food web and the factors influencing uptake and trophic transfer.

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risk in watersheds. However, not all AGC bioexposures occur via the diet and for many classes of AGCs, the predominant uptake pathway is direct uptake from water (Griscom and others, 2000; Croteau and Luoma, 2005; Luoma and Rainbow, 2005; Vajda and others, 2011; Barber and others, 2012). However, due to many compounds included in the AGC classification and the individual chemistry these compounds possess, the relative importance of direct vs. dietary uptake for many AGCs is unknown or depends on the taxon.

Assimilated contaminants can interfere with fundamental biological processes, which then trigger toxic effects (Rainbow and Luoma, 2011). For example, the extent to which fish experience toxic effects depends on the duration of exposure, life stage of exposure (that is, maternal, larval, and so forth) (Johnson and others, 2020), whether a contaminant is assimilated or lost at this level of the food web, and interactions with other environmental stressors (that is, temperature, disease, food availability, and so forth).

These broad factors influence bioexposures of both anthropogenic and geogenic contaminants but the specific understanding and status of knowledge for these two contaminant classes can vary widely based on their chemistry and sources as discussed below.

Anthropogenic Contaminants

It has been well established that mixtures of anthropogenic chemicals are ubiquitous in urban streams and pose a potential threat to water quality (Kolpin and others, 2002; Focazio and others, 2008; Bradley and others, 2017; Bradley and others, 2018). The major drivers of increased future risks to aquatic organisms from AGCs include changes in regional climates (for example, more droughts and wildfires, loss of wetland habitats), changes in land-use needs (for example, urbanization, new crop production), and increasing human populations with increasing water use (Van Metre and others, 2019). There are research opportunities to develop or adopt water quality and toxicity screening tools for the prioritization of “at-risk” stream reaches or watersheds, and quantitatively link AGC concentrations to current effects, and model effects as a function of AGC use and concentrations (Fischer and others, 2017; Barber and others, 2019; Van Metre and others, 2019; Barber and others, 2022; Zhang and others, 2023). This can lead to better hydrologic model predictions and assessment tools for water resource managers to improve infrastructure and best practices to maintain water quality for many generations to come.

Anthropogenic organic chemicals that enter the aquatic environment through wastewater point-source and through non-point sources come in a wide variety of physicochemical properties. These chemicals have different controlling mechanisms for their environmental fate and transport and risks for toxicity or other adverse biological effects once they enter the aquatic environment or drinking water. Moreover, many organic chemicals are often purposely designed to be biologically active (for example, pharmaceuticals and pesticides), or disruptive to biological functions and cell membranes (for example, surfactants, disinfectants, peroxides), or resist biological degradation (for

example, pharmaceuticals, per- and polyfluoroalkyl substances [PFAS], antioxidants, corrosion inhibitors). Some consumer, agricultural, and industrial products contain a mixture of chemicals having all these characteristics. The complexity of organic contaminants and aqueous mixtures is a challenge for summarizing collective effects on aquatic health. This challenge can be overcome by comprehensive site-based measurements of water chemistry and CECs and investigations of the exposure effects on the microbial and biological communities that exist and are exposed to this same complex mixture of chemicals (that is, simultaneous measurements and metrics). It is also important to fill in gaps in knowledge regarding bio-uptake rates and biomagnification of certain anthropogenic CECs. Moreover, strategic categorization of organic chemicals according to predominant sources or biological modes of action could promote faster assessment of overall risk and trend analyses to support decision making by water resource managers.

Geogenic Contaminants

Geogenic contaminants include metals (that is, cadmium [Cd], lead [Pb], Hg, zinc [Zn], manganese [Mn], chromium [Cr], and copper [Cu]) and metalloids (that is, arsenic [As], selenium [Se], antimony [Sb], and molybdenum [Mo]) that are concentrated and enriched in aquatic ecosystems by natural as well as anthropogenic activities that utilize geologic resources to meet core economic needs of society. Examples of such activities include the installation of drinking water wells, hard rock mining, oil refining, agricultural irrigation, and the construction of dams (Chapman and others, 2010). Wastewater discharge also contributes significant trace elements and metal contaminants into streams due to their use as purposeful constituents or impurities in consumer products (for example, Cr or cobalt [Co] in dyes, Zn or titanium [Ti] in sunscreens, As in household pesticides) or pharmaceuticals (for example, bismuth [Bi] in antibiotics, lithium [Li] in antidepressants). Drivers such as fire, climate (that is, precipitation), and hazards (that is, flooding), along with disturbance from human activities further enhance weathering and mobilization of geogenic contaminants from the landscape and delivery via dissolved and particulate forms to aquatic environments.

While TMDLs are focused on loads, those loads need to be converted into concentrations because bioavailability and toxicity effects to aquatic organisms are related to concentration. Indeed, EPA criteria are most often in the form of concentrations for which toxicity benchmarks and thresholds are based. Metal and metalloid bioexposure to aquatic organisms has been described by total concentrations (that is, the sum of all chemical species) or using thermodynamic models such as the free ion activity model (FIAM) and (or) the biotic ligand model (BLM). Both models assume the only bioavailable form of metals is the aqueous free ion, assuming direct and linear relationships between free ions and biological surface-bound metal species (Zhao and others, 2016; Mebane and others, 2020). However, these models do not account for uptake and exposure pathways beyond direct uptake of metal ions and biotic ligand complexation.

Total water column concentrations of geogenic contaminants alone are rarely reliable predictors of bioexposure, toxicity risk, and ecosystem health. Chemical speciation is an important factor when considering toxicity, bioavailability, and bioaccumulation across food webs. According to the International Union of Pure and Applied Chemistry (IUPAC) definition, chemical speciation refers to a specific form of an element defined as to isotopic composition, electronic or oxidations state, and (or) complex or molecular structure. By this definition, several changes in speciation can occur in the water column. For example, seasonal changes in dissolved O concentrations can not only induce oxidation state changes in geogenic contaminants but also changes in dissolved organic carbon (DOC) speciation and subsequent interactions with metals and biota. Transition metal (iron [Fe] and Mn) (oxyhydr)oxides and oxides can undergo reductive dissolution in anoxic environments, leading to the release of adsorbed metals and metalloids. Geogenic contaminants such as Cr, Hg, As, Se, and Sb can all be present with multiple oxidation states in aquatic environments and usually have a particular oxidation state that is more bioavailable and toxic compared to others. For example, As(III) is more toxic compared to As(V) (Spehar and others, 1980), but the opposite is true of Cr. In its reduced state, Cr(III) is generally less soluble and less toxic compared to the oxidized state Cr(VI) (Velma and others, 2009). Beyond changes in oxidation state, redox conditions also influence biological activity that can alter metal toxicity. For example, under anoxic conditions, As and Hg can undergo metabolic processes that produce toxic, methylated species, the latter of which bioaccumulates strongly across food webs (Brigham and others, 2009; Selin, 2009). Other metal C species that may form depending on redox conditions are organometallic complexes (Velma and others, 2009). Dissolved organic carbon speciation and metal complexing capacity are correlated in part to redox conditions, in that the abundance and quality of C will be altered depending on biological activity. The toxicity and bioavailability of these metal-carbon complexes are poorly understood.

Dissolved inorganic contaminants can also change from aqueous to solid species because of the precipitation of mineral phases. Chalcophilic metals and metalloids such as Hg, Zn, Pb, As and Cd can precipitate as sulfidic phases in the environment, especially under reducing conditions. Precipitation of these phases can radically decrease bioavailability and potential for bioaccumulation across food webs; however, if shifts in geochemical conditions occur, dissolution of the phases may lead to renewed bioavailability.

Physical and chemo-physical processes can also influence contaminant speciation and partitioning on to particles. Processes such as perturbation from benthic invertebrates or turbulence and scouring during high flow events can lead to higher concentrations of particulate matter in the water column that can adsorb geogenic contaminants. Particulate-adsorbed geogenic contaminants can have multiple effects on bioexposures. They can result in contaminant sequestration, if the complex is not biologically available, enhanced uptake via dietary consumption, transport, or burial via sedimentation processes. Yet, these processes also

depend on the characteristics of the particles present in the environment, including grain size and organic content, which can vary significantly in space and time ((Brigham and others, 2009; Stewart and others, 2008).

Whether a geogenic substance is an essential element further influences bioexposure. In the case of metal cation contaminants, microorganisms generally exhibit two primary water uptake mechanisms: (1) nonspecific binding to cell surfaces and biomass, and (2) metabolism-dependent intracellular uptake (Gadd and Griffiths, 1977; Stewart and others, 2010). Adsorption of metal cations to biomass is far more common, as metabolism-dependent intracellular transport is not only an energetically costly process but also can lead to the precipitation of intracellular metal phases, requiring the expression of tolerance mechanisms (Gadd and Griffiths, 1977). Uptake of the essential element Se, which forms oxyanions in the aqueous environment, is predominantly through an energy-mediated process (Stewart and others, 2004, 2010). For this reason, the partitioning of Se on inorganic particles in sediments tends to be low relative to its absorption by algae, making particle characterization necessary when trying to discern differences in Se associated with the particulate fraction both within and among aquatic systems (Ponton and others, 2020).

Trophic transfer of geogenic contaminants is highly dependent on trophic-specific and species-specific assimilation and loss (Reinfelder and others, 1998). For example, methylmercury (MeHg) is biomagnified in food webs because the proportion of MeHg to inorganic Hg increases with each trophic level and the loss of MeHg by an organism is relatively slow (Lavoie and others, 2013). Alternatively, studies have shown that Cd is biomagnified within distinct invertebrate and fish food webs, with invertebrate food webs having higher trophic enrichment factors than fish food webs (Croteau and others, 2005). In the case of Se, large differences in Se uptake have been quantified among algal and invertebrate taxa resulting in corresponding large differences along trophic pathways in Se concentrations in consumers and predators (Baines and Fisher, 2001; Stewart and others, 2004).

Priority Gaps in Anthropogenic and Geogenic Bioexposures

Table C1 presents a summary of the bioexposure knowledge gap analysis for ecosystem health and function. More details about these knowledge gaps and approaches to fill the gaps are presented in this section.

Anthropogenic Wastewater Contaminants of Emerging Concern Gaps

In a recent synthesis paper by Abbott and others (2019) comparing 464 water-cycle diagrams from around the world, it was estimated that approximately 24,000 cubic kilometers per year ($\text{km}^3 \cdot \text{yr}^{-1}$) is now used for human freshwater appropriation for livestock, croplands, forestry, industrial and domestic water use, and water necessary to dilute human pollutants (wastewater

use). This human use redistributes the equivalent of half of the global river streamflow and double the global groundwater recharge each year (Abbott and others, 2019). Despite this fact, it was determined that only 15 percent of the water-cycle diagrams represented human interaction with the water cycle, and only 2 percent showed climate change or water pollution effects on the water availability and quality. This is an important gap in water budget and water quality modeling and the USGS is well equipped to improve assessments of water security and sustainability by integrating human water use and indirect reuse into our hydrologic models. Human water use, reuse, and returned flows have become a major component of the global water cycle, resulting in significant contributions of additional nutrients, salinity shifts, and complex mixtures of CECs added to our Nation's surface water and groundwater (Kolpin and others, 2002; Abbott and others, 2019; Bradley and others, 2017, 2018; Barber and others, 2019, 2022). Treated domestic and industrial wastewater discharge, agricultural/irrigation and urban stormwater return flows, and chemical spills are all large contributors of these complex mixtures of CECs along with their associated adverse effects on water quality and ecosystem health downstream (Kolpin and others, 2002; Barber, 2014; Bradley and others, 2016; Fischer and others, 2017; Croteau and others, 2005, 2016; Masoner and others, 2019; Battaglin and others, 2020). With increasing and ever-changing mixtures of CECs being added to the natural aqueous environment through wastewater return flows, it is imperative for humans to better understand the adverse effects on human and ecosystem health and recreational sustainability, including the major drivers of these bioexposures and sources of priority pollutants.

Gap 1. Major Factors Affecting Human and Aquatic Organism Health

Wastewater Discharge and Toxicity Benchmark Data

At present there is no consolidated database of contaminant loads from the wide range of wastewater effluents discharged to natural waters. The scientific literature has many studies that investigate CECs detected in municipal WWTP effluent and sludge. This is largely facilitated by the fact that most municipal WWTP facilities are publicly owned treatment works (POTWs) and have publicly accessible information available through discharge permits. Some of these data for major point source discharges, including WWTPs, can be found using EPA's Enforcement and Compliance History Online (ECHO) tool (<https://echo.epa.gov/>) and EPA's Clean Watersheds Needs Survey (CWNS, <https://www.epa.gov/cwns>). However, other sources are often needed to obtain the treatment type and population served by the WWTPs, which allows for better fitting of chemical loads per capita and per treatment type to be used in modeling new watersheds. Other wastewater return-flow data are more difficult to obtain, such as those from industrial WWTP and mining discharges, as well as from non-point sources like agriculture or urban stormwater.

It is also difficult to obtain a consistent data set of aquatic toxicity benchmarks such as predicted no effect concentrations (PNECs), effect concentration at which 10 percent of a population is biologically affected (EC10), CMCs, or 50 percent lethality concentration (LC50) for the multitude organic contaminants used for pesticides, industrial applications, pharmaceuticals, or consumer products that are released into the aquatic environment (Brack and others, 2019). Similarly, a major data gap exists for bioaccumulation coefficients (BACs) or bioconcentration factors (BCFs) for priority pollutants; having this would help to better assess the health risk to aquatic organisms at different trophic levels, and potential risk to predator terrestrial organisms (including humans) that feed on aquatic organisms. Moreover, organic contaminants with high octanol-water coefficients (K_{ow}) can be an important driver to better understand which chemicals will bioaccumulate in fatty organs of the body like the liver or brain. Efforts to calculate, validate, or consolidate these BACs, BCFs, and toxicity benchmarks could form the basis for guidance documents on acceptable aqueous concentrations of CECs for different water use expectations. A collaborative effort by the USGS and other Federal agencies could provide the evidence-based scientific support and relevant toxicity benchmarks to stand up TMDLs for CECs currently lacking Federal regulation. USGS has particular expertise in addressing potential adverse biological effects caused by bioexposure of organisms to chemical loads associated with wastewater reuse.

Biogeochemical Model Parameters

Process-based models are only reliable with sufficient data for estimating parameters that will constrain the model simulations to perform accurate predictions. To be able to couple hydrologic and reactive transport models with ecological risk models, a deeper understanding of many of the biogeochemical processes in complex aqueous mixtures is needed to parameterize the models. Some essential parameters include chemical solubility, partition coefficients, complexing capabilities, and chemical degradation rates from biological or physical transformations in natural streams and WWTPs. The USGS is well-positioned to carry out laboratory-based and field-based investigations to identify processes controlling the transport and fate of anthropogenic organic contaminants in the aquatic environment, along with the possible exposure pathways to aquatic organisms and humans. Examples of existing USGS Water Mission Area (WMA) capabilities that could be expanded upon include: (1) laboratory exposure of aquatic snails to uranium (Croteau and others, 2016) and other contaminants of concern in the presence of different water chemistry conditions, and (2) continuous-flow mobile laboratories located on-site to test the endocrine or toxicity effects of wastewater effluent or agriculture runoff on fat head minnows (*Pimephales promelas*) (Barber and others, 2012, 2013, 2019) or other organisms in stream environments.

Gap 2. Water Quality Assessment Tools for Aquatic Organism and Human Health Risk

Water resource managers need better water quality assessment tools to sustain healthy streams and drinking water sources (Fischer and others, 2017). These tools include more affordable analytical measurements for CECs, access to CEC experts to assist with data interpretation, and production of hydrologic models that can simulate and predict stream concentrations of CECs in relationship to toxicity tolerance levels in a variety of climatic and water management scenarios. Owing to the variety of organic anthropogenic chemicals and the lack of sufficient empirical data for all their physicochemical properties and aquatic toxicology benchmarks, computation models could be used to supplement some of the information. These quantitative structure-activity relationship (QSAR) models use common molecular structures to estimate unknown physicochemical parameters to derive environmental fate, biodegradability, and potential toxicity tolerances of various organism types, are discussed in more detail in the “Develop Tools for Screening-Level Contaminant of Emerging Concern (CEC) Assessments” section.

Sustaining a healthy aquatic ecosystem for aquatic organisms, drinking water, and recreational use depends upon both water availability and water quality and requires scientifically informed water management decisions. Sufficient flow volumes are needed throughout the year by critical aquatic species to provide appropriate habitat space, aerated mixing, and maintain cool enough water temperatures. Water quality changes caused by anthropogenic activities and land use are also dominant drivers controlling the complex chemical mixtures that aquatic organisms are exposed to in streams, rivers, and lakes.

Bioactive anthropogenic chemicals released to the environment through spills and wastewater discharge pose direct toxicological and endocrine disruption risks to aquatic organisms (Omar and others, 2016). There are thousands of bioactive chemicals released into the aquatic environment from different product applications with a wide range of physicochemical parameters and ecotoxicology profiles (Kolpin and other, 2002). Water resource managers and regulators will continue to need scientific studies and new data to develop new tools and approaches for monitoring and modeling chemical concentrations, investigate mechanistic processes controlling fate and bio-exposure pathways of CECs, and provide interpretations to support water resource management decisions (Fischer and others, 2017). Filling these gaps will allow for the integration of chemical loading from major point-sources and non-point sources of anthropogenic return flows into the next generation of USGS water budget and reactive transport models to gain a more accurate water quality representation of our United States freshwater resources.

Develop Risk Prioritization Scheme

Water resource managers and citizens needing clean drinking water and desiring safe recreational opportunities can benefit from consistent, easy-to-use tools that assess the relative risks

and management opportunities for CECs in the Nation’s waters. Prioritization of the abundance of thousands of aquatic CECs with wide ranging sources, chemical classes, fate-controlling processes, and risk concerns makes it extremely challenging to carry out comprehensive and ongoing monitoring. To close this gap, the USGS could develop or adopt a consistent, evidence-based schema for prioritization of at-risk watersheds and high priority anthropogenic and geogenic CECs at the national and regional scale for different geographical and hydrological watershed types. There are existing risk prioritization schemes from USGS and others looking at stream concentrations in relationship to PNECs; persistence, bioaccumulation, toxicity, environmental occurrence (PBTO indices); regulatory pressure; risk perception by the public; along with other risk criteria and decision support systems that can be adapted to meet USGS stakeholder needs (Ankley and others, 2010; Ortiz de García and others, 2013; Nowell and others, 2014; Ortiz de García and others, 2014; Gramatica and others, 2016; Sangion and Gramatica, 2016; Card and others, 2017; Fischer and others, 2017; Covert and others, 2020; Medlock Kakaley and others, 2020; Barber and others, 2022; Faunce and others, 2023).

There are also publicly available databases that can be used to assist with compiling some of the data needed, including the EPA’s Estimation Parameter Interface (EPI) Suite (EPA, 2012; Card and others, 2017) and CompTox chemical dashboard (EPA, 2016a), the Aquatic Life Benchmarks and Ecological Risk Assessments for Registered Pesticides (EPA, 2021a), PubChem database (Kim and others, 2020), and PPDB—Pesticide Properties databases (Agriculture & Environment Research Unit [AERU], 2021). In addition, the EPA maintains a database of registered pesticides that have documented acute (that is, lethal concentration to kill 50 percent of a population [LC50] and half maximal efficiency concentration [EC50]) and chronic aquatic life benchmarks (ALBs) in fish, invertebrates, and aquatic plants (EPA, 2021a). It is much more difficult to find empirical measurements in the literature of these ALBs for non-pesticide CECs. The EPA has also established national recommendations for aquatic life criteria (criterion maximum concentrations [CMCs] and criterion continuous concentrations [CCCs]) for around 60 organic and inorganic toxic chemicals (EPA, 2021b). However, this list is mainly limited to nationally regulated chemicals and lacks criterion for thousands of other toxic and bioactive chemicals that need to have toxicological endpoints determined. The USGS could help fill this data gap by conducting a systematic collection and common databasing of toxicity endpoint criteria for aquatic organisms (for example, PNEC, environmental quality standard [EQS], no observed effect concentration [NOEC], lowest observed effect concentration [LOEC], EC50, LC50, and so forth) for a prioritized list of chemicals analogous to comprehensive aquatic toxicity databases (Norman and others, 2018) and tables in supporting information of Barber and others (2022). Examples include EPA’s maximum contaminant levels [MCLs] (<https://www.epa.gov/ground-water-and-drinking-water/national-primary-drinking-water-regulations>), EPA Human Health Benchmarks for Pesticides (<https://www.epa.gov/sdwa/2021-human-health-benchmarks-pesticides>), and

USGS Health-Based Screening Levels (<https://www.usgs.gov/programs/environmental-health-program/science/usgs-health-based-screening-levels-available-online>), along with a method for normalizing these criteria to allow cross comparisons for hazardous rankings (Norman and others, 2018). A similar open-access database could address physicochemical and biological uptake properties as well, starting with curating empirical data from the literature and credible databases.

Develop Tools for Screening-Level Contaminant of Emerging Concern Assessments

The development of (1) proxy-chemical analyses, (2) assimilation of computational toxicology and (3) predictive modeling for CECs would allow the USGS to provide screening-level assessments to support water resource decisions and to guide future monitoring and research studies in Next Generation Water Observing Systems (NGWOS) and IWS basins.

8. Proxy-chemical analyses: Spending time and resources to develop rapid, screening-level analytical methods that analyze for surrogate chemicals and metabolites across a range of chemical classes or source types could save money on costly analysis of larger suites of chemicals until after “contaminant hot spots” are identified. To make this analysis rapid and cost-effective, it cannot be all-inclusive and so finding suitable surrogate chemicals for entire chemical classes/source types and (or) other chemical proxies will be necessary. The combined scientific expertise of the USGS along with nation-wide access to NGWOS and IWS water basins allows for the USGS to use an “Integrated Design Approach” to investigate prioritized CEC effects and controlling processes. The ideal approach is to integrate multiple disciplines (for example, organic geochemists, hydrologists, biologists, and modelers) into NGWOS and IWS basin studies from the beginning.
9. Assimilation of computational toxicology: Due to the lack of empirical data for environmental fate parameters, biological uptake mechanisms, and toxicity endpoint criteria for the thousands of chemicals released from pharmaceuticals, consumer products, urban homes, and transportation systems into the environment, computational tools are an efficient way to screen large sets of chemicals in a short time with the potential for ranking the most environmentally hazardous constituents. The computational tools applied to organic chemical contaminants often rely on QSAR models. These are regression models that use structural molecular descriptors within organic chemicals and commonly measured fate parameters like water solubility and octanol/water partition coefficient (K_{ow}) to predict environmental fate parameters and biological activity of chemicals with similar descriptors. For example, the K_{ow} is important for assessing organic contaminant fate and transport, because hydrophobic compounds ($\log K_{ow} > 3$) preferentially accumulate in solid phases and hydrophilic compounds

($\log K_{ow} < 3$) preferentially occur in the aqueous phase. QSAR tools like those found in EPA’s EPI Suite interface are being used to help make predictions of environmental concentrations of CECs and produce hazardous ranking indexes like (i) PNEC_{equivalents} and Risk Quotients (Barber and others, 2022; Faunce and others, 2023), (ii) apply PBTO indices to rank persistence, bioaccumulation, toxicity, and environmental occurrence (Ortiz de García and others, 2013; Sangion and Gramatica, 2016), or (iii) provide a software tool for Decision Analysis by Ranking Techniques (DART; Pavan and Worth, 2008a & 2008b) to support the ranking of chemical according to their environmental and toxicological concern.

10. Predictive modeling for CECs: The USGS accumulated wastewater ratio (ACCWW) on-line mapper is a versatile tool which can combine empirical data inputs (when available) with QSAR-based inputs (EPA, 2012 & 2016) coupled with streamflow, wastewater discharge, and ecotoxicity models to produce geographic information system (GIS) maps of watersheds and stream reaches showing predicted environmental concentrations of CECs (Kandel and others, 2017; Faunce and others, 2023; Barber and others, 2022). This provides a powerful screening-level assessment tool for regional watersheds that can be scaled up to national level or scaled down to local level. Innovative platforms for presenting water availability, water quality, and water suitability on GIS maps can improve the speed and accessibility of science data delivered. The supporting information of Barber and others (2022) provides a large data table of pharmaceuticals, personal care products (PPCP), and pesticides ecotoxicity benchmarks (LC50, EC50, PNECs, NOECs) from literature-based empirical data and EPI Suite ECOSAR model predictions, and also includes an accumulated wastewater model for the Shenandoah watershed with a risk prioritization scheme using water quality data.

Geogenic Contaminant Gaps

Gap 3. Mechanistic Understanding of Uptake Dynamics of Geogenic Contaminants for Watershed Assessments of Risk and Coupled Hydrologic-Biogeochemical Model Development

A key gap in understanding how geogenic contaminants result in bioexposures that affect ecosystem health are the factors and drivers that control the availability of geogenic contaminants in water and particulate material (Luomo and Rainbow, 2009). Specifically, what environmental conditions favor dissolved chemical species of metals that are readily adsorbed onto particles or directly taken up by microorganisms, algae, or plants? Then, once associated with particles (sediments or organic material), what types of particles favor high assimilation of metals by aquatic species? Given that many

metals, including metalloids, like As and Se, show the largest enrichment at the base of aquatic food webs, from water onto particles, understanding of these fundamental processes could lead to significant improvements in evaluating bioexposures as part of integrated assessments (Stewart and others, 2010).

The complexity of resolving these processes across systems can be daunting. In the absence of a more mechanistic approach, scientists have utilized distribution coefficients (K_d), a ratio that quantifies the partitioning of dissolved forms of metals onto particles. The K_d ratio is defined by the concentration of the constituent in particulate phase divided by the concentration in water. Over the past several decades this ratio has been used to evaluate bioavailability of mined metals (Stewart and Malley, 1999) and extensively in the “Ecosystem Selenium Model” developed by Presser and Luoma (2010) and for developing EPA site-specific criteria for Se. In the case of Se, partitioning inferred from K_d values was found to vary over several orders of magnitude among systems (freshwater to marine), which could be attributed in part to Se species (that is, selenate, selenite, selenomethionine), residence times, and particle types (that is, sediment, biofilms, filamentous algae) (Presser and Luoma, 2010). While K_d ratios for Se may be effective in identifying and even predicting gross differences in partition of Se among systems, they often lack the resolution within watersheds to estimate differences in bioavailable Se at spatially and temporally relevant scales. Moreover, without the mechanistic basis underlying the differences in K_d , it is not possible for resource managers to identify specific factors controlling bioexposures in food webs for the development of mitigation strategies. Mechanistic processes are also required for building coupled hydrologic-biogeochemical models for anticipating changes in bioavailable Se and other metalloids and metals in response to climate and resource management actions. A mechanistic approach to assessing and predicting changes in geogenic bioexposures would include understanding: (1) biogeochemical (that is, redox, pH, DOC, sulfate, and so forth) and hydrologic (that is, mixing, residence time) controls on chemical speciation, (2) potential for different chemical species to be adsorbed and (or) accumulated by different particles (that is, sediment, microorganisms, algae, and plants), and (3) relative assimilation of aqueous versus particulate chemical species by organisms. Given their inter-dependent nature, these processes could be evaluated jointly to determine their net effect in different aquatic environments.

The USGS can investigate these processes across a range of geochemical and hydrological conditions, for example, initially in relevant IWS basins such as the Upper Colorado River as well as other basins with outstanding Se and related metal or metalloid problems such as the Kootenai River (Idaho and Montana). Selenium and other geogenic metalloids and metals derived from mining remain a concern for resource managers in these areas (EPA, 2017) and there are ongoing research efforts (such as transboundary projects) that could provide historical and ongoing data collections to support the process research (Chapman and others, 2010). This effort could later be extended to the other IWS basins (that is, the Delaware River and Illinois River Basins) as appropriate to validate processes and model applications.

Biogeochemical Controls on Chemical Speciation

This approach uses field based water quality collections of total and dissolved geogenic species and ancillary parameters (that is, redox, pH, DOC, sulfate, nutrients, suspended solids, and so forth) combined with speciation modeling (Visual Minteq, <https://vminteq.lwr.kth.se/>; and PHREEQC, <https://www.usgs.gov/software/phreeqc-version-3>) (Parkhurst and Appelo, 2013). The goal is to determine how measured chemical speciation changes with geochemical conditions and test if those changes can be estimated using geochemical speciation models. Speciation can be defined as oxidation state, phase change (for example, adsorption to particulate matter, precipitation as a solid) and complexation with dissolved organic compounds. Sites could be chosen along geochemical gradients and hydrologic residence times. Geogenic elements could be selected for this work based on WMA, Federal partners, state agencies, and stakeholder priorities.

Chemical Speciation Controls on Particulate Adsorption and Bioaccumulation

This approach determines how chemical species (quantified above) are adsorbed or accumulated by different particle types that can influence their bioavailability. Quantifying particle characteristics in nature and corresponding aqueous geochemical conditions can enhance understanding of the controls that particles exert on metal speciation. This entails characterization of particulate material and geogenic particulate concentrations from sites described previously. Particulate material (greater than 0.45 micrometer [μm]) will be characterized by: (1) physical characteristics (that is, particle size and concentration), (2) inorganic composition (that is, grain size, Fe, Si, aluminum [Al] content, and so forth), (3) organic matter content and composition (that is, percent carbon, chlorophyll a, stable isotopes $\delta^{13}\text{C}$, nitrogen-15 [$\delta^{15}\text{N}$], and sulfur-34 [$\delta^{34}\text{S}$] signatures), and (4) algal and biomass taxonomy. Chemical extractions, specific elemental analysis (CHNS analyzer), TOC analysis and stable isotope analysis are all example of initial characterization techniques. Flow fractionation inductively coupled plasma mass-spectrometry (ICP-MS) and spectrometric methods offer other analytical approaches for particulate matter characterization. Additionally, particle-bound metal speciation (for example, oxidation state, organo-metallic complexes, and so forth) can be characterized using advanced spectroscopic techniques such as X-ray adsorption spectroscopy (XAS). This technique will help answer mechanistic questions about metal speciation and associations with elements such as Fe and Al in particulates. The information can then be used to improve predictive geochemical models, better accounting for speciation and the influence of particulate matter on bioavailability. The knowledge of this technique resides within the Strategic Laboratory Science Branch (SLSB) of Laboratory and Analytical Services Division (LASD) in WMA. Typically, this technique is carried out at a synchrotron facility and the idea is that these measurements will be paired with in-house techniques (that is, chemical extraction, Fourier

Transform infrared spectroscopy, and so forth) to create a framework of understanding. However, benchtop XAS is possible and could be explored in the future.

Assimilation of Geogenic Contaminants from Inorganic and Organic Particles

This approach combines field assessments of dissolved chemical species and particulates and to determine their control on bioavailability to biota (that is, invertebrates or fish). Results from field measurements could be used to guide laboratory uptake experiments in the form of feeding studies. The feeding studies could utilize different particle and dissolved metal species, using well proven methods (Cain and others, 2016; Croteau and others, 2017).

Hydrologic Controls on Chemical Speciation and Particulate Bioavailability

Hydrological investigations can provide insight into how transport processes that affect residence times influence biogeochemical transformation and speciation, geogenic adsorption and bioaccumulation. Residence times can be estimated at field sites to aid in the evaluation of differences in bio-uptake among data from different locations.

Gap 4. Bioavailability and Toxicity of Organometallic Complexes

Another geogenic gap is bioavailability and toxicity of organometallic complexes. Models such as FIAM and BLM often underestimate metal and (or) metalloid bioexposure, not accounting for the bioavailability of organometallic complexes (Mebane and others, 2020). Natural dissolved organic matter (herein identified as DOM) is an example of a widespread environmental metal or metalloid complexing agent whose reactivity is overlooked by these classic thermodynamic models. DOM has varying degrees of reactivity toward metal/metalloids depending on chemical properties, such as molecular weight, chemical structure, and functionality (Cory and McKnight, 2005), and environmental properties such as DOM source and geochemical conditions (for example, pH, ionic strength, and redox conditions). For example, calcophile metals such as Cd, Pb, Hg, and Zn all bind strongly to thiol groups associated with DOM (Shindo and Brown, 1965; Chen and others, 2017; Lescord and others, 2018). Similarly, sulfidic groups associated with DOM have been shown to be important complexing agents of arsenite (Hoffman and others, 2013). However, changes in metal oxidation state as well as changes in DOM acidity and functionality influence metal complexation (Ren and others, 2017). Thus, predicting metal complexation with DOM is not always as simple as a straightforward calculation.

As complicated as the formation of metal- or metalloid-DOM complexes is, understanding the influence on bioavailability and uptake of these complexes adds another layer of complexity. The specific metal or metalloid, the type and chemical characteristics of DOM and the organism all influence organometallic complex

uptake and bioaccumulation across food webs. For example, Cd-uptake rates in Zebra mussels (*Dreissena polymorpha*) were 52–61 percent lower when complexed to humic acid (HA, a fraction of DOM) compared to the free ion but still higher than predicted by FIAM, with some Cd-HA accumulation in the organism (Voets and other, 2004). A study with zebra mussels demonstrated different extents of uptake of Cd, silver (Ag), and Hg depending on whether low molecular-weight or high molecular-weight DOM was present (Roditi and others, 2000). In the presence of HA, fulvic acid (FA) and different polysaccharides, the uptake of Pb by green algae was less than the free ion (Pb^{2+}) but still greater than FIAM-predicted values (Lamelas and others, 2005). Complexation with HA and FA lead to higher internalized concentrations of Pb compared to polysaccharides, demonstrating how the type of OC can influence the extent of uptake (Lamelas and others, 2005). A study in artificial sea water showed that Pb uptake and toxicity in marine invertebrates—blue mussels (*Mytilus edulis*) and sea urchins (*Paracentrotus lividus*)—actually increased when complexed with HA compared to the free Pb ion (Sánchez-Marín and others, 2007). The bioaccumulation of Cu also has been shown to increase in algae when high initial Cu concentrations are complexed with low molecular-weight FAs (Shi and others, 2018). Higher molecular-weight FAs have been shown to contain more aliphatic functional groups and less carboxyl groups per unit C mass (Tan and Giddens, 1972; Reemtsma and These, 2005; Shi and others, 2018), which makes it less susceptible for binding with cationic metals such as Cu.

The source of DOM affects molecular composition, affecting reactivity with metal or metalloids and subsequent complex bioavailability. DOM can be autochthonous, meaning it is derived from an aquatic ecosystem (that is, detritus, biomass, and so forth) or allochthonous, meaning it is terrestrially derived (that is, plants, soil, and so forth). Allochthonous DOM is generally considered more recalcitrant compared to autochthonous DOM, as heterotrophic and autotrophic metabolic processes generally lead to the production of small aliphatic acids, while plant and soil organic matter can be more aromatic and thus, typically considered less biodegradable (Thurman, 1985). Molecular composition of DOM varies spatially and temporally due to its proximity to source material and its exposure to environmental processes such as biodegradation and photodegradation (Helms and others, 2008; Hansen and others, 2016). DOM composition can also be influenced by anthropogenic activities. For example, DOM composition along a WWTP-river-lake continuum was shown to vary greatly, with the ratio of protein-like to humic-like carbon notably decreasing and the percentage of humic-like carbon increasing (EPA, 1999; Barber and others, 2001; Wang and others, 2019). Therefore, when considering the bioavailability of organometallic complexes, it is important to consider changes in DOM composition that may occur in the environment and how this could influence the extent of metal or metalloid complexation.

Bioexposure Studies of Metal-Dissolved Organic Matter Complexes with Organisms at the Base of the Food Web

This approach could involve field and laboratory-based experiments, which will lead to the development of analytical tools and the improvement of existing bioexposure models.

Experiments could vary with respect to four factors: (1) metal and (or) metalloids of interest, (2) DOM characteristics sources, (3) microbial species, and (4) analytical techniques. We suggest beginning studies by focusing on microbial species, as they are a key component of the base of the food web; however, there are many data gaps that exist for other aquatic species such as invertebrates. The intention is that studies could build in complexity, both from a geochemical and ecological standpoint.

Metal or metalloids of interest could be prioritized based on the needs of IWS such as the Upper Colorado, Delaware, and Illinois River Basins. This determination will involve reconnaissance of IWS basin aqueous geochemical conditions via field campaigns and analysis of available historical data. Field and lab measurements, as well as historical data, could be incorporated into speciation models, such as PHREEQC, to confirm and predict dominant metal or metalloid species, as discussed in the “Biochemical Controls on Chemical Speciation” section in this chapter.

During experiments, DOM from different sources with different chemical characteristics could be varied to evaluate differences in metal-DOM binding and effects on bioexposure. Sources of DOM will vary by basin environments and by environments affected by anthropogenic activity (that is, WWTP, urban, agricultural). Constraints could be imposed based on the importance or contribution to an environment of interest.

Microbial species are an essential part of the base of the food web composition and as such it is important to specifically understand the bio-exposure of metal-DOM complexes to these species. Beginning work could focus on microbial species to establish an understanding at the base of the food web and future studies could examine bioavailability to more complex organisms. Both pure culture and whole community experiments could be conducted in the laboratory. Cell numbers, growth rates and potentially genes of interest could be quantified in relation to metal-DOM complex exposure.

Laboratory-based experiments could be designed to measure metal-DOM complexes in new and novel ways. Ideal DOM sources such as Suwannee River fulvic acid could be used with metals to prepare metal-DOM complexes and chromatographic and spectroscopic techniques could be explored for detection and quantification of organometallic complexes.

Timelines

Table C2 presents a suggested timeline to conduct prioritized USGS approaches for closing knowledge gaps to address the anthropogenic and geogenic contaminant bioexposures affecting aquatic ecosystems outlined in this chapter.

Table C2. Timeline summary of prioritized U.S. Geological Survey (USGS) approaches to closing knowledge gaps for the anthropogenic and geogenic contaminant bioexposures affecting aquatic ecosystems.

[DNA, deoxyribonucleic acid; K_{oc} , carbon-water coefficient; K_{ow} , octanol-water coefficient; yr, year; IWS, Integrated Water Science; PNEC, predicted no-effect concentrations; EC50, half maximal efficiency concentration; CEC, contaminant of emerging concern; WWTP, wastewater treatment plant; QSAR, quantitative structure-activity relationship; DART, decision analysis by ranking techniques; GIS, geographic information system.]

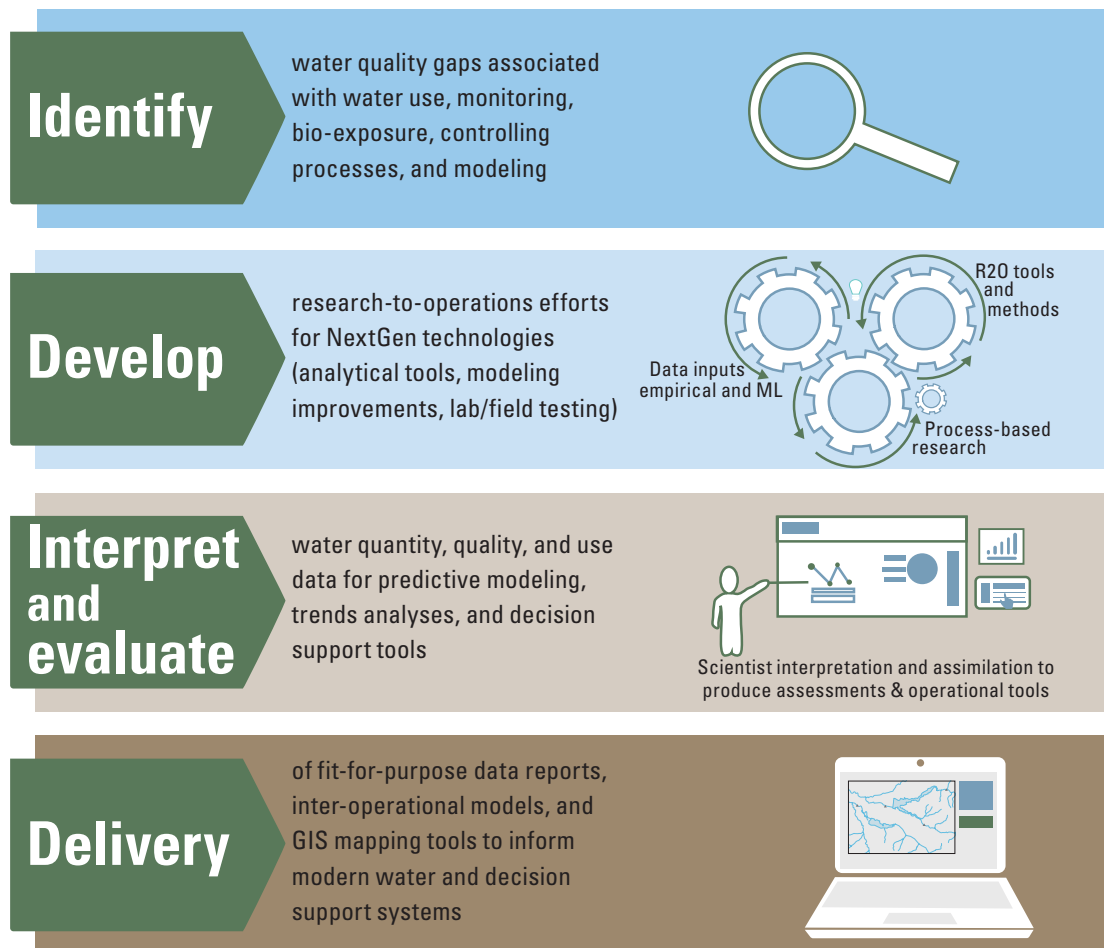
Advancements	Major processes affecting human and aquatic organism health	Water quality assessment tools for aquatic organism and human health risk implications	Mechanistic understanding of uptake dynamics of geogenic contaminants	Bioavailability and toxicity of organometallic complexes
Near-term (2-yr)	Organization and access to data for assessing indirect water reuse affecting biota. Focusing on physicochemical properties such as water solubility, K_{ow} , K_{oc} , and so forth, and toxicity values such as PNEC, EC50, bioaccumulation coefficients and bioconcentration factors.	Compilation of a well-rounded database of toxicity endpoint criteria. Generation of a list of key anthropogenic contaminants of concern.	Field collections and laboratory studies to identify key hydrologic-biogeochemical factors controlling uptake of geogenics.	Selection of metals of interest in relation to IWS basins. Field and laboratory characterization of organic matter. Initial microcosm (that is, exposure) studies.
Mid-term (2- to 5-yr)	Feeding experiments of organisms such as snails and fish, in complex mixtures of contaminants such as those in WWTP. In-situ mobile laboratory experiments using environmental complex mixtures.	Improvements to QSAR and DART modeling tools. Field studies and computational toxicity tools to investigate CECs and potential proxies that can be used to predict the risk of anthropogenic contaminants	Development and testing of a simple coupled hydrologic-biogeochemical model to predict geogenic bioavailability and uptake in focus systems.	Development of improved analytical techniques for organometallic complex detection. Bioaccumulation microcosm experiments.
Long-term (10-yr)	Results from database compilation, laboratory and field work systematically incorporated into improved biogeochemical process-based models throughout this 10-year period.	Linkages between ecotoxicology studies and improved models to GIS mapping tools.	Enhancement of particulate matter characterization (that is, e-DNA, hyperspectral imaging); development and testing of proxies for select metals and water quality constituents.	Detection of organometallic complexes, in-situ improvements to predictive models with respect to organometallic bioavailability and ecosystem effects

Expected Outcomes

Understanding the bio-exposure and uptake of AGC contributes to WMA priorities in water availability including quantity, quality, and support of beneficial uses and ecosystem health. Addressing questions surrounding these complex processes and mechanisms requires broad, interdisciplinary, cross-mission area approaches (fig. C2). The gaps in knowledge and approaches discussed here are not exhaustive but are considered some of the most pressing gaps to address to improve our predictive capabilities surrounding water quality. It is important to note that our recommendations can be coordinated with other objectives being developed by the WMA. Below are some key expected outcomes from this work:

- Identification of key metal species (as opposed to relying on total metal concentrations) and organic CECs contributing to bio-uptake and bioaccumulation concerns for aquatic organisms.
- Characterization of the speciation and bioavailability of metal and organic chemical associated particulate matter in IWS basins, improving predictive modeling.
- Determination of hydrologic-biogeochemical mechanisms contributing to residence time thresholds that play a key role in uptake of AGCs.
- Development of a framework to predict the bioavailability and uptake of organometallic complexes, based on metal and organic carbon sources unique to water bodies of concern, such as IWS basins.
- Improved analytical capabilities for rapid, cost-effective measurements of target organic chemical indicators indicative of various environmental stressors and anthropogenic sources.
- Improved analytical capabilities for measuring organometallic complexes. This type of measurement will be valuable to assess water quality and improve model inputs.
- Creation of a database of toxicity endpoint criteria for aquatic and human consumption for a prioritized list of chemicals, along with a method for normalizing these and providing a qualitative assessment tool associated with these toxicity values.
- Identification of key measurements that can be used as a proxy for anthropogenic contaminants to provide an alternative approach when discrete measurements of all contaminant types is cost prohibitive.
- Improvement of predictive modeling capabilities through the extensive development of QSAR and DART tools.
- Linkages between ecotoxicology models and GIS tools that can be used to assess water quality at variable (local to national) scales.

Figure C2. Diagram of a proposed U.S. Geological Survey (USGS) integrated interdisciplinary, cross-mission-area approach to deliver actionable water-availability science to address complex processes and mechanisms for solving knowledge gaps in anthropogenic and geogenic contaminant bioexposures affecting aquatic ecosystems. GIS, geographic information systems; ML, machine learning; and R2O, research to operation.



References Cited

- Abbott, B.W., Bishop, K., Zarnetske, J.P., Minaudo, C., Chapin, F.S., III, Krause, S., Hannah, D.M., Conner, L., Ellison, D., Godsey, S.E., Plont, S., Marçais, J., Kolbe, T., Huebner, A., Frei, R.J., Hampton, T., Gu, S., Buhman, M., Sayedi, S.S., Ursache, O., Chapin, M., Henderson, K.D., and Pinay, G., 2019, Human domination of the global water cycle absent from depictions and perceptions: *Nature Geoscience*, v. 12, no. 7, p. 533–540, <https://doi.org/10.1038/s41561-019-0374-y>.
- Agriculture & Environment Research Unit [AERU], 2021, PPDB—Pesticide properties database: Hertfordshire, U.K., University of Hertfordshire, Agriculture & Environment Research Unit [AERU], accessed November 1, 2021, at <https://sitem.herts.ac.uk/aeru/ppdb/en/>.
- Ankley, G.T., Bennett, R.S., Erickson, R.J., Hoff, D.J., Hornung, M.W., Johnson, R.D., Mount, D.R., Nichols, J.W., Russom, C.L., Schmieder, P.K., Serrano, J.A., Tietge, J.E., and Villeneuve, D.L., 2010, Adverse outcome pathways—A conceptual framework to support ecotoxicology research and risk assessment: *Environmental Toxicology and Chemistry*, v. 29, no. 3, p. 730–741, <https://doi.org/10.1002/etc.34>.
- Baines, S.B., and Fisher, N.S., 2001, Interspecific differences in the bioconcentration of selenite by phytoplankton and their ecological implications: *Marine Ecology Progress Series*, v. 213, p. 1–12, <https://doi.org/10.3354/meps213001>.
- Barber, L.B., 2014, Emerging Contaminants, chap. 1.13 of Ahuja, S., ed., *Comprehensive water quality and purification*: Waltham, Mass., Elsevier, p. 245–266.
- Barber, L.B., Keefe, S.H., Brown, G.K., Furlong, E.T., Gray, J.L., Kolpin, D.W., Meyer, M.T., Sandstrom, M.W., and Zaugg, S.D., 2013, Persistence and potential effects of complex organic contaminant mixtures in wastewater-impacted streams: *Environmental Science & Technology*, v. 47, no. 5, p. 2177–2188, <https://doi.org/10.1021/es303720g>.
- Barber, L.B., Leenheer, J.A., Noyes, T.I., and Stiles, E.A., 2001, Nature and transformation of dissolved organic matter in treatment wetlands: *Environmental Science & Technology*, v. 35, no. 24, p. 4805–4816, <https://doi.org/10.1021/es010518i>.
- Barber, L.B., Faunce, K.E., Bertolatus, D.W., Hladik, M.L., Jasmann, J.R., Keefe, S.H., Kolpin, D.W., Meyer, M.T., Rapp, J.L., Roth, D.A., and Vajda, A.M., 2022, Watershed-scale risk to aquatic organisms from complex chemical mixtures in the Shenandoah River: *Environmental Science and Technology*, v. 56, no. 2, p. 845–861, <https://doi.org/10.1021/acs.est.1c04045>.
- Barber, L.B., Rapp, J.L., Kandel, C., Keefe, S.H., Rice, J., Westerhoff, P., Bertolatus, D.W., and Vajda, A.M., 2019, Integrated assessment of wastewater reuse, exposure risk, and fish endocrine disruption in the Shenandoah River watershed: *Environmental Science & Technology*, v. 53, no. 7, p. 3429–3440, <https://doi.org/10.1021/acs.est.8b05655>.
- Barber, L.B., Vajda, A.M., Douville, C., Norris, D.O., and Writer, J.H., 2012, Fish endocrine disruption responses to a major wastewater treatment facility upgrade: *Environmental Science & Technology*, v. 46, no. 4, p. 2121–2131, <https://doi.org/10.1021/es202880e>.
- Barnes, K.K., Kolpin, D.W., Furlong, E.T., Zaugg, S.D., Meyer, M.T., and Barber, L.B., 2008, A national reconnaissance of pharmaceuticals and other organic wastewater contaminants in the United States—I) Groundwater: *Science of the Total Environment*, v. 402, nos. 2–3, p. 192–200, <https://doi.org/10.1016/j.scitotenv.2008.04.028>.
- Battaglin, W., Duncker, J., Terrio, P., Bradley, P., Barber, L., and DeCicco, L., 2020, Evaluating the potential role of bioactive chemicals on the distribution of invasive Asian carp upstream and downstream from river mile 278 in the Illinois waterway: *Science of the Total Environment*, v. 735, article 139458, p. XX, <https://doi.org/10.1016/j.scitotenv.2020.139458>.
- Brack, W., Aissa, S.A., Backhaus, T., Dulio, V., Escher, B.I., Faust, M., Hilscherova, K., Hollender, J., Hollert, H., Müller, C., and Munthe, J., 2019, Effect-based methods are key—The European Collaborative Project SOLUTIONS recommends integrating effect-based methods for diagnosis and monitoring of water quality: *Environmental Sciences Europe*, v. 31, no. 1, 6 p., <https://doi.org/10.1186/s12302-019-0192-2>.
- Bradley, P.M., Journey, C.A., Button, D.T., Carlisle, D.M., Clark, J.M., Mahler, B.J., Nakagaki, N., Qi, S.L., Waite, I.R., and Van Metre, P.C., 2016, Metformin and other pharmaceuticals widespread in wadeable streams of the southeastern United States: *Environmental Science & Technology Letters*, v. 3, no. 6, p. 243–249, <https://doi.org/10.1021/acs.estlett.6b00170>.
- Bradley, P.M., Journey, C.A., Romanok, K.M., Barber, L.B., Buxton, H.T., Foreman, W.T., Furlong, E.T., Glassmeyer, S.T., Hladik, M.L., Iwanowicz, L.R., Jones, D.K., Kolpin, D.W., Kuivila, K.M., Loftin, K.A., Mills, M.A., Meyer, M.T., Orlando, J.L., Reilly, T.J., Smalling, K.L., and Villeneuve, D.L., 2017, Expanded target-chemical analysis reveals extensive mixed-organic-contaminant exposure in U.S. streams: *Environmental Science & Technology*, v. 51, no. 9, p. 4792–4802, <https://doi.org/10.1021/acs.est.7b00012>.
- Bradley, P.M., Kolpin, D.W., Romanok, K.M., Smalling, K.L., Focazio, M.J., Brown, J.B., Cardon, M.C., Carpenter, K.D., Corsi, S.R., DeCicco, L.A., Dietze, J.E., Evans, N., Furlong, E.T., Givens, C.E., Gray, J.L., Griffin, D.W., Higgins, C.P., Hladik, M.L., Iwanowicz, L.R., Journey, C.A., Kuivila, K.M., Masoner, J.R., McDonough, C.A., Meyer, M.T., Orlando, J.L., Strynar, M.J., Weis, C.P., and Wilson, V.S., 2018, Reconnaissance of mixed organic and inorganic chemicals in private and public supply tapwaters at selected residential and workplace sites in the United States: *Environmental Science & Technology*, v. 52, no. 23, p. 13972–13985, <https://doi.org/10.1021/acs.est.8b04622>.

- Brigham, M.E., Wentz, D.A., Aiken, G.R., and Krabbenhoft, D.P., 2009, Mercury cycling in stream ecosystems, 1—Water column chemistry and transport: *Environmental Science & Technology*, v. 43, no. 8, p. 2720–2725, <https://doi.org/10.1021/es802694n>.
- Cain, D.J., Croteau, M.-N., Fuller, C.C., and Ringwood, A.H., 2016, Dietary uptake of Cu sorbed to hydrous iron oxide is linked to cellular toxicity and feeding inhibition in a benthic grazer: *Environmental Science & Technology*, v. 50, no. 3, p. 1552–1560, <https://doi.org/10.1021/acs.est.5b04755>.
- Card, M.L., Gomez-Alvarez, V., Lee, W.-H., Lynch, D.G., Orentas, N.S., Lee, M.T., Wong, E.M., and Boethling, R.S., 2017, History of EPI Suite and future perspectives on chemical property estimation in US Toxic Substances Control Act new chemical risk assessments: *Environmental Science, Processes & Impacts*, v. 19, no. 3, p. 203–212, <https://doi.org/10.1039/C7EM00064B>.
- Chapman, P.M., Adams, W.J., Brooks, M.L., Delos, C.G., Luoma, S.N., Maher, W.A., Ohlendorf, H.M., Presser, T.S., Shaw, D.P., eds., 2010, Ecological assessment of selenium in the aquatic environment: Boca Raton, CRC Press, 358 p.
- Chen, H., Johnston, R.C., Mann, B.F., Chu, R.K., Tolic, N., Parks, J.M., and Gu, B., 2017, Identification of mercury and dissolved organic matter complexes using ultrahigh resolution mass spectrometry: *Environmental Science & Technology Letters*, v. 4, no. 2, p. 59–65, <https://doi.org/10.1021/acs.estlett.6b00460>.
- Cory, R.M., and McKnight, D.M., 2005, Fluorescence spectroscopy reveals ubiquitous presence of oxidized and reduced quinones in dissolved organic matter: *Environmental Science & Technology*, v. 39, no. 21, p. 8142–8149, <https://doi.org/10.1021/es0506962>.
- Covert, S.A., Shoda, M.E., Stackpoole, S.M., and Stone, W.W., 2020, Pesticide mixtures show potential toxicity to aquatic life in U.S. streams, water years 2013–2017: *Science of the Total Environment*, v. 745, article 141285, 12 p., <https://doi.org/10.1016/j.scitotenv.2020.141285>.
- Croteau, M.-N., Cain, D.J., and Fuller, C.C., 2017, Assessing the dietary bioavailability of metals associated with natural particles—Extending the use of the reverse labeling approach to zinc: *Environmental Science & Technology*, v. 51, no. 5, p. 2803–2810, <https://doi.org/10.1021/acs.est.6b06253>.
- Croteau, M.-N., Fuller, C.C., Cain, D.J., Campbell, K.M., and Aiken, G., 2016, Biogeochemical controls of uranium bioavailability from the dissolved phase in natural freshwaters: *Environmental Science & Technology*, v. 50, no. 15, p. 8120–8127, <https://doi.org/10.1021/acs.est.6b02406>.
- Croteau, M.-N., and Luoma, S.N., 2005a, Delineating copper accumulation pathways for the freshwater bivalve *Corbicula* using stable copper isotopes: *Environmental Toxicology and Chemistry*, v. 24, no. 11, p. 2871–2878, <https://doi.org/10.1897/04-608R.1>.
- Croteau, M.-N., Luoma, S.N., and Stewart, A.R., 2005b, Trophic transfer of metals along freshwater food webs—Evidence of cadmium biomagnification in nature: *Limnology and Oceanography*, v. 50, no. 5, p. 1511–1519, <https://doi.org/10.4319/lo.2005.50.5.1511>.
- de Paiva Magalhães, D., da Costa Marques, M.R., Baptista, D.F., and Buss, D.F., 2015, Metal bioavailability and toxicity in freshwaters: *Environmental Chemistry Letters*, v. 13, no. 1, p. 69–87, <https://doi.org/10.1007/s10311-015-0491-9>.
- Eagles-Smith, C.A., Suchanek, T.H., Colwell, A.E., Anderson, N.L., and Moyle, P.B., 2008, Changes in fish diets and food web mercury bioaccumulation induced by an invasive planktivorous fish: *Ecological Applications*, v. 18, no. sp8, p. A213–A226, <https://doi.org/10.1890/06-1415.1>.
- Faunce, K.E., Keefe, S.H., Barber, L.B., Jasmann, J.R., and Rapp, J.L., 2023, Modeling wastewater reuse and risk posed by wastewater-derived contaminants in the Potomac River Watershed: *Journal of the American Water Resources Association*, article JAWR-22-0077-P, 24 p., <https://doi.org/10.1111/1752-1688.13110>.
- Fischer, A., Ter Laak, T., Bronders, J., Desmet, N., Christoffels, E., van Wezel, A., and van der Hoek, J.P., 2017, Decision support for water quality management of contaminants of emerging concern: *Journal of Environmental Management*, v. 193, p. 360–372, <https://doi.org/10.1016/j.jenvman.2017.02.002>.
- Focazio, M.J., Kolpin, D.W., Barnes, K.K., Furlong, E.T., Meyer, M.T., Zaugg, S.D., Barber, L.B., and Thurman, M.E., 2008, A national reconnaissance for pharmaceuticals and other organic wastewater contaminants in the United States—II) Untreated drinking water sources: *Science of the Total Environment*, v. 402, nos. 2–3, p. 201–216, <https://doi.org/10.1016/j.scitotenv.2008.02.021>.
- Gadd, G.M., and Griffiths, A.J., 1977, Microorganisms and heavy metal toxicity: *Microbial Ecology*, v. 4, no. 4, p. 303–317, <https://doi.org/10.1007/BF02013274>.
- Gramatica, P., Cassani, S., and Sangion, A., 2016, Aquatic ecotoxicity of personal care products—QSAR models and ranking for prioritization and safer alternatives' design: *Green Chemistry*, v. 18, no. 16, p. 4393–4406, <https://doi.org/10.1039/C5GC02818C>.
- Griscom, S.B., Fisher, N.S., and Luoma, S.N., 2000, Geochemical influences on assimilation of sediment-bound metals in clams and mussels: *Environmental Science & Technology*, v. 34, no. 1, p. 91–99, <https://doi.org/10.1021/es981309+>.
- Hansen, A.M., Kraus, T.E.C., Pellerin, B.A., Fleck, J.A., Downing, B.D., and Bergamaschi, B.A., 2016, Optical properties of dissolved organic matter (DOM)—Effects of biological and photolytic degradation: *Limnology and Oceanography*, v. 61, no. 3, p. 1015–1032, <https://doi.org/10.1002/lno.10270>.

- Helms, J.R., Stubbins, A., Ritchie, J.D., Minor, E.C., Kieber, D.J., and Mopper, K., 2008, Absorption spectral slopes and slope ratios as indicators of molecular weight, source, and photobleaching of chromophoric dissolved organic matter: *Limnology and Oceanography*, v. 53, no. 3, p. 955–969, <https://doi.org/10.4319/lo.2008.53.3.0955>.
- Hoffmann, M., Mikutta, C., and Kretzschmar, R., 2013, Arsenite binding to natural organic matter—Spectroscopic evidence for ligand exchange and ternary complex formation: *Environmental Science & Technology*, v. 47, no. 21, p. 12165–12173, <https://doi.org/10.1021/es4023317>.
- Johnson, R.C., Stewart, A.R., Limburg, K.E., Huang, R., Cocherell, D., and Feyrer, F., 2020, Lifetime chronicles of selenium exposure linked to deformities in an imperiled migratory fish: *Environmental Science & Technology*, v. 54, no. 5, p. 2892–2901, <https://doi.org/10.1021/es4023317>.
- Kandel, C.M., Rapp, J.L., and Barber, L.B., 2017, Shenandoah River Accumulated Wastewater Ratio Mapper: U.S. Geological Survey data release. [Superseded by ver.2.0, February 2019, which is available at <https://doi.org/10.5066/F7RF5S8X>.]
- Kim, S., Chen, J., Cheng, T., Gindulyte, A., He, J., He, S., Li, Q., Shoemaker, B.A., Thiessen, P.A., Yu, B., Zaslavsky, L., Zhang, J., and Bolton, E.E., 2020, PubChem in 2021—New data content and improved web interfaces: *Nucleic Acids Research*, v. 49, no. D1, p. D1388–D1395, <https://doi.org/10.1093/nar/gkaa971>.
- Kolpin, D.W., Furlong, E.T., Meyer, M.T., Thurman, E.M., Zaugg, S.D., Barber, L.B., and Buxton, H.T., 2002, Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. streams, 1999–2000—A national reconnaissance: *Environmental Science & Technology*, v. 36, no. 6, p. 1202–1211, <https://doi.org/10.1021/es011055j>.
- Lamelas, C., Wilkinson, K.J., and Slaveykova, V.I., 2005, Influence of the composition of natural organic matter on Pb bioavailability to microalgae: *Environmental Science & Technology*, v. 39, no. 16, p. 6109–6116, <https://doi.org/10.1021/es050445t>.
- Lavoie, R.A., Jardine, T.D., Chumchal, M.M., Kidd, K.A., and Campbell, L.M., 2013, Biomagnification of mercury in aquatic food webs—A worldwide meta-analysis: *Environmental Science & Technology*, v. 47, no. 23, p. 13385–13394, <https://doi.org/10.1021/es403103t>.
- Lepak, R.F., Krabbenhoft, D.P., Ogorek, J.M., Tate, M.T., Bootsma, H.A., and Hurley, J.P., 2015, Influence of *Cladophora*–quagga mussel assemblages on nearshore methylmercury production in Lake Michigan: *Environmental Science & Technology*, v. 49, no. 13, p. 7606–7613, <https://doi.org/10.1021/es506253v>.
- Lepak, J.M., Robinson, J.M., Kraft, C.E., and Josephson, D.C., 2009, Changes in mercury bioaccumulation in an apex predator in response to removal of an introduced competitor: *Ecotoxicology*, v. 18, no. 5, p. 488–498, <https://doi.org/10.1007/s10646-009-0306-5>.
- Lescord, G.L., Emilson, E.J.S., Johnston, T.A., Branfireun, B.A., and Gunn, J.M., 2018, Optical properties of dissolved organic matter and their relation to mercury concentrations in water and biota across a remote freshwater drainage basin: *Environmental Science & Technology*, v. 52, no. 6, p. 3344–3353, <https://doi.org/10.1021/acs.est.7b05348>.
- Luoma, S.N., and Rainbow, P.S., 2009, Metal contamination in aquatic environments—Science and lateral management: *Integrated Environmental Assessment and Management*, v. 5, no. 3, p. 492, <https://doi.org/10.1897/1551-3793-5.3.492>.
- Masoner, J.R., Kolpin, D.W., Cozzarelli, I.M., Barber, L.B., Burden, D.S., Foreman, W.T., Forshay, K.J., Furlong, E.T., Groves, J.F., Hladik, M.L., Hopton, M.E., Jaeschke, J.B., Keefe, S.H., Krabbenhoft, D.P., Lowrance, R., Romanok, K.M., Rus, D.L., Selbig, W.R., Williams, B.H., and Bradley, P.M., 2019, Urban stormwater—An overlooked pathway of extensive mixed contaminants to surface and groundwaters in the United States: *Environmental Science & Technology*, v. 53, no. 17, p. 10070–10081, <https://doi.org/10.1021/acs.est.9b02867>.
- Mebane, C.A., Chowdhury, M.J., De Schampelaere, K.A.C., Loftis, S., Paquin, P.R., Santore, R.C., and Wood, C.M., 2020, Metal bioavailability models—Current status, lessons learned, considerations for regulatory use, and the path forward: *Environmental Toxicology and Chemistry*, v. 39, no. 1, p. 60–84, <https://doi.org/10.1002/etc.4560>.
- Medlock Kakaley, E.K., Blackwell, B.R., Cardon, M.C., Conley, J.M., Evans, N., Feifarek, D.J., Furlong, E.T., Glassmeyer, S.T., Gray, L.E., Hartig, P.C., Kolpin, D.W., Mills, M.A., Rosenblum, L., Villeneuve, D.L., and Wilson, V.S., 2020, De facto water reuse—Bioassay suite approach delivers depth and breadth in endocrine active compound detection: *Science of the Total Environment*, v. 699, article 134297, 12 p., <https://doi.org/10.1016/j.scitotenv.2019.134297>.
- Norman, J.E., Toccalino, P.L., and Morman, S.A., 2018, Health-based screening levels for evaluating water-quality data (2d ed.): U.S. Geological Survey web page, <https://doi.org/10.5066/F71C1TWP>.
- Nowell, L.H., Norman, J.E., Moran, P.W., Martin, J.D., and Stone, W.W., 2014, Pesticide toxicity index—A tool for assessing potential toxicity of pesticide mixtures to freshwater aquatic organisms: *Science of the Total Environment*, v. 476–477, p. 144–157, <https://doi.org/10.1016/j.scitotenv.2013.12.088>.

- Omar, T.F.T., Ahmad, A., Aris, A.Z., and Yusoff, F.M., 2016, Endocrine disrupting compounds (EDCs) in environmental matrices—Review of analytical strategies for pharmaceuticals, estrogenic hormones, and alkylphenol compounds: *TrAC Trends in Analytical Chemistry*, v. 85, p. 241–259, <https://doi.org/10.1016/j.trac.2016.08.004>.
- Ortiz de García, S.A., Pinto, G.P., García-Encina, P.A., and Irusta-Mata, R., 2013, Ranking of concern, based on environmental indexes, for pharmaceutical and personal care products—An application to the Spanish case: *Journal of Environmental Management*, v. 129, p. 384–397, <https://doi.org/10.1016/j.jenvman.2013.06.035>.
- Ortiz de García, S.A., Pinto, G.P., García-Encina, P.A., and Irusta-Mata, R., 2014, Ecotoxicity and environmental risk assessment of pharmaceuticals and personal care products in aquatic environments and wastewater treatment plants: *Ecotoxicology*, v. 23, no. 8, p. 1517–1533, <https://doi.org/10.1007/s10646-014-1293-8>.
- Parkhurst, D.L., and Appelo, C.A.J., 2013, Description of input and examples for PHREEQC version 3—A computer program for speciation, batch-reaction, one-dimensional transport, and inverse geochemical calculations: U.S. Geological Survey Techniques and Methods book 6, chap. A43, 497 p., <https://doi.org/10.3133/tm6A43>.
- Pavan, M., and Worth, A., 2008a, A set of case studies to illustrate the applicability of DART (decision analysis by ranking techniques) in the ranking of chemicals: Joint Research Center Scientific and Technical Reports, no. EUR 23481 EN—2008, 82 p., accessed November 1, 2021 at https://publications.jrc.ec.europa.eu/repository/bitstream/JRC47007/applicability%20of%20dart_eur%20report_final.pdf.
- Pavan, M., and Worth, A.P., 2008b, Publicly-accessible QSAR software tools developed by the Joint Research Centre: SAR and QSAR in Environmental Research, v. 19, nos. 7–8, p. 785–799, <https://doi.org/10.1080/10629360802550390>.
- Ponton, D.E., Graves, S.D., Fortin, C., Janz, D., Amyot, M., and Schiavon, M., 2020, Selenium interactions with algae—Chemical processes at biological uptake sites, bioaccumulation, and intracellular metabolism: *Plants*, v. 9, no. 4, special issue, article 528, 26 p., <https://doi.org/10.3390/plants9040528>.
- Presser, T.S., and Luoma, S.N., 2010, A methodology for ecosystem-scale modeling of selenium: *Integrated Environmental Assessment and Management*, v. 6, no. 4, p. 685–710, <https://doi.org/10.1002/ieam.101>.
- Rainbow, P.S., and Luoma, S.N., 2011, Metal toxicity, uptake and bioaccumulation in aquatic invertebrates—Modelling zinc in crustaceans: *Aquatic Toxicology*, v. 105, nos. 3–4, p. 455–465, <https://doi.org/10.1016/j.aquatox.2011.08.001>.
- Reemtsma, T., and These, A., 2005, Comparative investigation of low-molecular-weight fulvic acids of different origin by SEC-Q-TOF-MS—New insights into structure and formation: *Environmental Science & Technology*, v. 39, no. 10, p. 3507–3512, <https://doi.org/10.1021/es0480466>.
- Reinfelder, J.R., Fisher, N.S., Luoma, S.N., Nichols, J.W., and Wang, W.-X., 1998, Trace element trophic transfer in aquatic organisms—A critique of the kinetic model approach: *Science of the Total Environment*, v. 219, nos. 2–3, p. 117–135, [https://doi.org/10.1016/S0048-9697\(98\)00225-3](https://doi.org/10.1016/S0048-9697(98)00225-3).
- Ren, J., Fan, W., Wang, X., Ma, Q., Li, X., Xu, Z., and Wei, C., 2017, Influences of size-fractionated humic acids on arsenite and arsenate complexation and toxicity to *Daphnia magna*: *Water Research*, v. 108, p. 68–77, <https://doi.org/10.1016/j.watres.2016.10.052>.
- Roditi, H.A., Fisher, N.S., and Sañudo-Wilhelmy, S.A., 2000, Uptake of dissolved organic carbon and trace elements by zebra mussels: *Nature*, v. 407, article 6800, p. 78–80, <https://doi.org/10.1038/35024069>.
- Sánchez-Marín, P., Lorenzo, J.I., Blust, R., and Beiras, R., 2007, Humic acids increase dissolved lead bioavailability for marine invertebrates: *Environmental Science & Technology*, v. 41, no. 16, p. 5679–5684, <https://doi.org/10.1021/es070088h>.
- Sangion, A., and Gramatica, P., 2016, PBT assessment and prioritization of contaminants of emerging concern: Pharmaceuticals: *Environmental Research*, v. 147, p. 297–306, <https://doi.org/10.1016/j.envres.2016.02.021>.
- Schlekat, C.E., Lee, B.-G., and Luoma, S.N., 2002, Assimilation of selenium from phytoplankton by three benthic invertebrates—Effect of phytoplankton species: *Marine Ecology Progress Series*, v. 237, p. 79–85, <https://doi.org/10.3354/meps237079>.
- Selin, N.E., 2009, Global Biogeochemical Cycling of Mercury—A review: *Annual Review of Environment and Resources*, v. 34, no. 1, p. 43–63, <https://doi.org/10.1146/annurev.environ.051308.084314>.
- Shi, W., Fang, X., Wu, X., Zhang, G., Que, W., and Li, F., 2018, Alteration of bioaccumulation mechanisms of Cu by microalgae in the presence of natural fulvic acids: *Chemosphere*, v. 211, p. 717–725, <https://doi.org/10.1016/j.chemosphere.2018.07.200>.
- Shindo, H., and Brown, T.L., 1965, Infrared spectra of complexes of L-cysteine and related compounds with zinc(II), cadmium(II), mercury(II), and lead(II): *Journal of the American Chemical Society*, v. 87, no. 9, p. 1904–1909, <https://doi.org/10.1021/ja01087a013>.
- Spehar, R.L., Fiandt, J.T., Anderson, R.L., and DeFoe, D.L., 1980, Comparative toxicity of arsenic compounds and their accumulation in invertebrates and fish: *Archives of Environmental Contamination and Toxicology*, v. 9, no. 1, p. 53–63, <https://doi.org/10.1007/BF01055499>.

- Stewart, A.R., Grosell, M., Buchwalter, D.B., Fisher, N.S., Luoma, S.N., Matthews, T., Orr, P., and Wang, W.-X., 2010, Bioaccumulation and trophic transfer of selenium, chap 5 of Chapman, P.M., Adams, W.J., Brooks, M.L., Delos, C.G., Luoma, S.N., Maher, W.A., Ohlendorf, H.M., Presser, T.S., and Shaw, P.D., eds., *Ecological assessment of selenium in the aquatic environment*: Boca Raton, Flor., CRC Press, p. 93–140.
- Stewart, A.R., Luoma, S.N., Schlekot, C.E., Doblin, M.A., and Hieb, K.A., 2004, Food web pathway determines how selenium affects aquatic ecosystems—A San Francisco Bay case study: *Environmental Science & Technology*, v. 38, no. 17, p. 4519–4526, <https://doi.org/10.1021/es0499647>.
- Stewart, A.R., and Malley, D.F., 1999, Effect of metal mixture (Cu, Zn, Pb, and Ni) on cadmium partitioning in littoral sediments and its accumulation by the freshwater macrophyte *Eriocaulon septangulare*: *Environmental Toxicology and Chemistry*, v. 18, no. 3, p. 436–447, <https://doi.org/10.1002/etc.5620180311>.
- Stewart, A.R., Saiki, M.K., Kuwabara, J.S., Alpers, C.N., Marvin-DiPasquale, M., and Krabbenhoft, D.P., 2008, Influence of plankton mercury dynamics and trophic pathways on mercury concentrations of top predator fish of a mining-impacted reservoir: *Canadian Journal of Fisheries and Aquatic Sciences*, v. 65, no. 11, p. 2351–2366, <https://doi.org/10.1139/F08-140>.
- Tan, K.H., and Giddens, J.E., 1972, Molecular weights and spectral characteristics of humic and fulvic acids: *Geoderma*, v. 8, no. 4, p. 221–229, [https://doi.org/10.1016/0016-7061\(72\)90013-4](https://doi.org/10.1016/0016-7061(72)90013-4).
- Tesoriero, A.J., Erickson, M.L., Conaway, C.H., Tomaszewski, E.J., and Green, C.T., 2024, Knowledge gaps and opportunities for understanding water-quality processes affecting water availability for beneficial uses: U.S. Geological Survey Open-File Report 2023–1086, 81 p., <https://doi.org/10.3133/ofr20231086>.
- Thurman, E.M., 1985, Organic geochemistry of natural waters, part of *Developments in Biogeochemistry*, v. 2: Springer Dordrecht, 497 p., available at <https://doi.org/10.1007/978-94-009-5095-5>.
- U.S. Environmental Protection Agency [EPA], 1999, Protocol for developing sediment TMDLs (1st ed.): U.S. Environmental Protection Agency, Office of Water, no. EPA-841-B-99-004, 143 p. [Available from <https://www.epa.gov/nscep>.]
- U.S. Environmental Protection Agency [EPA], 2012, Estimation program interface (EPI) Suite for Microsoft Windows (4.11 ed.): U.S. Environmental Protection Agency software release, accessed November 1, 2023, at <https://www.epa.gov/tsca-screening-tools/download-epi-suite-estimation-program-interface-v411>.
- U.S. Environmental Protection Agency [EPA], 2016, Computational toxicology chemicals dashboard (CompTox dashboard): U.S. Environmental Protection Agency database, accessed November 1, 2021, at <https://comptox.epa.gov/>.
- U.S. Environmental Protection Agency [EPA], 2017, National Summary of State Information: U.S. Environmental Protection Agency, accessed January 2021, at https://iaspub.epa.gov/waters10/attains_nation_cy.control. [Also available at <https://www.epa.gov/waterdata/attains>]
- U.S. Environmental Protection Agency [EPA], 2021a, Aquatic life benchmarks and ecological risk assessments for registered pesticides: U.S. Environmental Protection Agency database, accessed November 1, 2021, at <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-and-ecological-risk>.
- U.S. Environmental Protection Agency [EPA], 2021b, National recommended water quality criteria—Aquatic life criteria table: U.S. Environmental Protection Agency database, accessed November 1, 2021, at <https://www.epa.gov/wqc/national-recommended-water-quality-criteria-aquatic-life-criteria-table>.
- Vajda, A.M., Barber, L.B., Gray, J.L., Lopez, E.M., Bolden, A.M., Schoenfuss, H.L., and Norris, D.O., 2011, Demasculinization of male fish by wastewater treatment plant effluent: *Aquatic Toxicology*, v. 103, nos. 3–4, p. 213–221, <https://doi.org/10.1016/j.aquatox.2011.02.007>.
- Van Metre, P.C., Waite, I.R., Qi, S., Mahler, B., Terando, A., Wiczorek, M., Meador, M., Bradley, P., Journey, C., Schmidt, T., and Carlisle, D., 2019, Projected urban growth in the southeastern USA puts small streams at risk: *PLOS ONE*, v. 14, no. 10, article e0222714, 17 p., <https://doi.org/10.1371/journal.pone.0222714>.
- Velma, V., Vutukuru, S.S., and Tchounwou, P.B., 2009, Ecotoxicology of hexavalent chromium in freshwater fish—A critical review: *Reviews on Environmental Health*, v. 24, no. 2, p. 129–146, <https://doi.org/10.1515/REVEH.2009.24.2.129>.
- Voets, J., Bervoets, L., and Blust, R., 2004, Cadmium bioavailability and accumulation in the presence of humic acid to the zebra mussel, *Dreissena polymorpha*: *Environmental Science & Technology*, v. 38, no. 4, p. 1003–1008, <https://doi.org/10.1021/es034742e>.
- Wang, W.X., Fisher, N.S., and Luoma, S.N., 1996, Kinetic determinations of trace element bioaccumulation in the mussel *Mytilus edulis*: *Marine Ecology Progress Series*, v. 140, p. 91–113, <https://doi.org/10.3354/meps140091>.
- Wang, Y., Hu, Y., Yang, C., Wang, Q., and Jiang, D., 2019, Variations of DOM quantity and compositions along WWTPs-river-lake continuum—Implications for watershed environmental management: *Chemosphere*, v. 218, p. 468–476, <https://doi.org/10.1016/j.chemosphere.2018.11.037>.
- Weisman, R. J., Barber, L. B., Rapp, J. L., and Ferreira, C. M., 2019, De facto reuse and disinfection by-products in drinking water systems in the Shenandoah River watershed: *Environmental Science: Water Research & Technology*, v. 5, no. 10, p. 1699–1708, <https://doi.org/10.1039/C9EW00326F>.

Zhang, L., Zhang, Y., Zhu, M., Chen, L., and Wu, B., 2023, A critical review on quantitative evaluation of aqueous toxicity in water quality assessment: *Chemosphere*, article no. 140159, 12 p., <https://doi.org/10.1016/j.chemosphere.2023.140159>.

Zhao, C.-M., Campbell, P.G.C., and Wilkinson, K.J., 2016, When are metal complexes bioavailable?: *Environmental Chemistry*, v. 13, no. 3, p. 425–433, <https://doi.org/10.1071/EN15205>.