

Stoichiometric Ecotoxicology for a Multisubstance World

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Abstract

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Nutritional and contaminant stressors influence organismal physiology, trophic interactions, community structure, and ecosystem-level processes; however, the interactions between toxicity and elemental imbalance in food resources have been examined in only a few ecotoxicity studies. Integrating well-developed ecological theories that cross all levels of biological organization can enhance our understanding of ecotoxicology. In the present article, we underline the opportunity to couple concepts and approaches used in the theory of ecological stoichiometry (ES) to ask ecotoxicological questions and introduce stoichiometric ecotoxicology, a subfield in ecology that examines how contaminant stress, nutrient supply, and elemental constraints interact throughout all levels of biological organization. This conceptual framework unifying ecotoxicology with ES offers potential for both empirical and theoretical studies to deepen our mechanistic understanding of the adverse outcomes of chemicals across ecological scales and improve the predictive powers of ecotoxicology.

Keywords : nutrient ratios, elemental imbalances, toxicity, ecotoxicological models, multiple stressors

1 **Introduction**

2 Ecotoxicology is the study of the harmful effects of chemical contaminants on
3 individual organisms, populations, communities, and ecosystem processes (Cairns Jr
4 1988; Baird et al. 1996; Calow 1996; Relyea and Hoverman 2006; Gessner and Tlili
5 2016; Walker, Sibly, and Peakall 2016). Ecotoxicologists have found that toxicity may be
6 influenced by environmental factors such as ambient physico-chemical conditions
7 influencing contaminant bioavailability, organismal traits (e.g., differential
8 metabolism/sensitivity, trophic guild), and/or food webs or ecosystem structure (Newman
9 and Clements 2007). While these findings are often incorporated into regulatory actions,
10 many ‘big-picture’ ecological theories that cross all levels of biological organization have
11 not been consistently integrated into the field of ecotoxicology and subsequent
12 applications during chemicals assessment and management.

13 How organisms obtain the nutrition and energy needed for maintenance, growth,
14 and reproduction is a central concept in many ecological studies that examine organismal
15 physiology (Wagner et al. 2013), food webs (Phillips and Eldridge 2006), biodiversity
16 (Worm et al. 2002), and biogeochemical cycles (Sterner and Elser 2002). Given how
17 nutrition can be an important driver in many ecological studies, it is no surprise that
18 multiple different frameworks are used to examine how diets affect organismal
19 phenotypes. All nutritional ecological frameworks in the broadest sense examine how
20 nutrition affects the physiology, behavior, life-history, and eco-evolutionary responses of
21 organisms and their interactions with the environment (Raubenheimer, Simpson, and
22 Mayntz 2009). Frameworks such as optimal foraging and nutritional geometry have a
23 behavioral ecology foundation that focuses on optimizing fitness traits based on food

1 selection. Initially, nutritional geometry focused on controlled laboratory experiments to
2 examine the optimal nutrition of two macronutrients (i.e., proteins, carbohydrates). More
3 recent techniques have including adding a third nutrient to investigate more complex
4 diets using the right-angle triangle (Raubenheimer 2011); however, much of the work has
5 focused on organisms and is not readily scaled to communities and ecosystems. Other
6 frameworks such as ecological stoichiometry (ES) that tracks elements and have a
7 foundational focus on ecosystem science (Sturner and Elser 2002; Sperfeld et al. 2017).
8 Early work in this area focused on interspecific competition and biogeochemical cycles
9 (e.g., Elser et al. 1998) whereas more recent work includes a focus on the physiological
10 changes that occur due to element-limited diets (Wagner et al. 2013). While ES is limited
11 partly because some types of micronutrient limitation (i.e., essential fatty acids/amino
12 acids) are not easily identified by studying elements alone, recent work has extended the
13 ES modeling framework to incorporate concepts from nutritional geometry and has
14 shown that macronutrients can be used as currencies (Anderson et al. 2020). Overall, the
15 relative ease of tracking elements through food webs that can span all spatial levels of
16 organization from genes to ecosystems (Hessen et al. 2013) makes ES an ideal
17 framework to examine how toxicants and elemental stress affect organisms and
18 ecosystems.

19 At the most fundamental level, ES is the study of imbalances between the
20 elemental composition of available resources and what is required for organismal
21 metabolism (Sturner and Elser 2002). In essence, imbalances between multiple available
22 elements and organismal requirements of those elements strongly affect the physiology of
23 many taxa and result in widely variable growth, reproduction, and survival (Frost et al.

1 2005; Wagner et al. 2013; Sterner and Elser 2002). Stoichiometric effects on organisms
2 in turn alter ecosystems and biogeochemical cycles (Elser et al. 1998). If the organism
3 elemental requirements are imbalanced with the elemental supply ratios, it would alter
4 the nutrition of the organism and effect their excretion elemental ratios. For example, an
5 imbalanced nitrogen:phosphorus (N:P) supply ratios can alter the nutrition of
6 zooplankton consumers and their N and P excretion ratios (Elser and Urabe 1999).
7 ES examines the causes and consequences of elemental imbalances across biological
8 levels by linking cellular processes and organismal metabolism to population dynamics,
9 predator-prey interactions, and ecosystem function (Sterner and Elser 2002).

10 As elemental ratios in resources vary widely and elemental limitation is
11 widespread in nature, ecotoxicology may benefit from the incorporation of a
12 stoichiometric perspective. Traditionally, toxicity experiments use nutrient-replete
13 conditions, with the intention to eliminate nutrient limitation and elemental imbalances,
14 because of the inherent assumption that nutrient limitation would interact with toxins or
15 toxicants to alter results. While nutrient replete conditions may not be commonplace in
16 natural ecosystems, only a few ecotoxicology studies have examined toxicity-elemental
17 imbalance interactions. When explicitly included in single-species toxicity testing, the
18 effect of the contaminant has been found to vary when there are elemental imbalances
19 between the test organism and its nutritional resources (e.g., Hansen et al. 2008; Fulton et
20 al. 2009; Fulton et al. 2010; Lessard and Frost 2012; Bian et al. 2013; Conine and Frost
21 2017). While this handful of examples demonstrate the possible uses of ES to improve
22 our understanding of ecotoxicology, this past work represents only a small part of what
23 could be a larger conceptually united subdiscipline: stoichiometric ecotoxicology. We

1 define stoichiometric ecotoxicology as a subfield in ecology that examines how
2 imbalances between organisms and their food resources interact with contaminant stress
3 in all levels of biological organization. Further, we define nutrients in this framework as
4 elements that are conserved across all spatial scales from genes to ecosystems.

5 One area of stoichiometric ecotoxicology that could be especially useful but
6 remains undeveloped is the application of formal mathematical approaches that couple
7 mass balance constraints to ecological processes. Stoichiometric models track multiple
8 elements to understand how elemental imbalances affect processes from cells to whole
9 ecosystems across multiple levels of biological organization. For example, at the
10 organism scale, mass-balance models couple information on physiological processes of
11 multiple elements (ingestion and assimilation) to determine animal elemental demands
12 and the optimal nutritional mixture needed for maximal growth (Sterner 1997; Frost and
13 Elser 2002; Frost et al. 2006). In addition, organismal processes are incorporated in
14 dynamical models that quantitatively link energy flow and elemental cycling to predator-
15 prey interactions (Loladze, Kuang, and Elser 2000; Elser et al. 2012; Muller et al. 2001).
16 These models could be extended to examine the ecological effects of natural or
17 anthropogenic contaminants by modifying parameters relative to chemical concentration
18 or body burden. Similarly, classical ecotoxicological models can be expanded to
19 incorporate stoichiometric constraints by modifying parameters related to nutrient
20 limitations. Integrative modeling approaches would allow for examination of interactive
21 effects of nutrients and contaminants on organismal physiological state, competitive
22 interactions, and whole-ecosystem mass balance. Additionally, these models may help
23 determine critical contaminant concentration thresholds that regulate biomass, growth

1 rates, ecosystem function, and the biological transfer of toxins through ecosystems
2 (Andersen, Elser, and Hessen 2004; Evans-White et al. 2009; Suzuki-Ohno, Kawata, and
3 Urabe 2012). Current models are beginning to incorporate stoichiometric constraints to
4 understand the ecological effects of chemical toxicity ((Peace, Poteat, and Wang 2016;
5 Hassan et al. 2018), as discussed below) and this effort calls for conceptual, theoretical,
6 and empirical development.

7 Here we examine opportunities to develop stoichiometric ecotoxicology by
8 coupling concepts and approaches commonly used in ES to ask ecotoxicology questions.
9 In particular, we present a conceptual overview of the links between elemental
10 stoichiometry and toxicology, discuss recent advances in stoichiometric ecotoxicology
11 research, and explore possible areas of future development, including the development
12 and application of new or modified mathematical models. Altogether we show the need
13 and promise of stoichiometric ecotoxicology to provide the impetus for its future
14 development.

15 16 **The conceptualization of stoichiometric ecotoxicology**

17 While the potential value of examining interactions between stoichiometry and
18 toxicology has been noted (see above), a next step is to further develop the empirical
19 basis of stoichiometric ecotoxicology through the study of a diverse range of organisms, a
20 broader range of toxic contaminants, and with more types of elemental imbalances. The
21 underlying rationale and unifying concept of this research field is that the supply of
22 multiple resources can alter or be altered by the effects of chemical toxicity on
23 organisms, populations, communities, and ecosystems (Figure 1). As further development
24 of this field will require examining these connections using a combination of empirical

1 and theoretical approaches, a starting point is to identify the possible types and nature of
2 interactions between toxins and toxicants and unbalanced elemental supply. In addition,
3 this perspective is not taxa- or trophic-level specific, as we envision it being applied to
4 toxicological research on all organisms including producers, consumers, detritivores, and
5 predators. Here we provide an initial overview of stoichiometric ecotoxicology and its
6 potential applications across the various levels of biological organization (Figure 1).
7 Also, we have compiled a list of studies that implicitly or explicitly investigate the role of
8 stoichiometric constraints on the response of organisms to diverse toxic substances,
9 including pesticides, pharmaceuticals, essential and nonessential metals, and natural
10 substances such as plant secondary compounds (Table 1). These studies represent a
11 starting point from which we identify the types of ecotoxicological questions that a
12 stoichiometric framework allows us to ask at multiple levels of biological organization.
13
14 **Organismal stoichiometry and ecotoxicology.** Organismal physiology and life-history
15 are strongly affected by both available food resources (Figure 1, arrow 2) and
16 contaminants (Figure 1, arrow 3), and these alterations to metabolism may interact to
17 create more variation in toxicity. The nature of these interactive effects largely depends
18 on the nutrients under consideration, the organismal traits, and the nature of the
19 contaminant. For example, similar to the concentration addition mechanism of
20 contaminant mixtures (Altenburger et al. 1996), if a toxicant and elemental stress affect
21 the same metabolic pathways it could lead to an additive or synergistic stress response.
22 Additionally, exposure to toxicants may activate detoxification pathways within an
23 organism that can involve the synthesis of compounds rich in carbon (C) or other
24 elements. If these detoxification compounds mainly involve C, they may help alleviate

1 elemental imbalances through disposing of excess C supplies that accompany elemental
2 imbalances. Alternatively, excessive C demands could eventually lead to energy
3 limitation and reduced nutrient use efficiency. If these detoxification requirements
4 involve a non-C element in short supply, toxicity could exacerbate elemental limitation or
5 impair repair in exposed organisms.

6 The presence of contaminants may also directly alter the nutritional quality of
7 food resources available to both primary producers and consumers (Figure 1, arrow 1).
8 Numerous metals produce non-soluble complexes with elements that are important
9 nutritionally, such as P. For example, aluminum and phosphate form non-soluble
10 complexes, and so the presence of aluminum may reduce P supply and lead to skewed
11 N:P ratios available to primary producers. This could result in increased P- limitation of
12 organismal growth and altered ecological processes (Kopáček et al. 2000; Clivot et al.
13 2014). Indeed, aluminum additions have been used to intentionally reduce P availability
14 in lakes for some time, despite its known toxicity (e.g., Steinman and Ogdahl 2008).

15 Contaminants may also directly alter the C:N:P ratios of producers and consumers
16 by altering their physiology (e.g., via increased maintenance costs associated with
17 recovery from toxin exposure; Figure 1 arrow 3). Das et al. (2014) found reduced C:P
18 and N:P content of phytoplankton communities exposed to silver nanoparticles, while
19 Cherchi et al. (2015) found opposite trends in the cyanobacteria, *Anabaena variabilis*,
20 exposed to nano-titanium dioxide. Fulton et al. (2010) working on the aquatic
21 macrophyte, *Lemna gibba*, showed that triclosan (a common antimicrobial agent) caused
22 changes in tissue C:P ratios, depending on the N:P ratio of the culture medium. Exposure
23 to ionic silver has also been shown to reduce leaf litter microbial conditioning, resulting

1 in higher leaf litter C:N and C:P ratio (Arce-Funck et al. 2013). Nutrient stoichiometry
2 has also been shown to alter the toxicity of contaminants to plants (Fulton et al. 2009;
3 Fulton et al. 2010; Mkandawire and Dudel 2012; Bian et al. 2013; Brooks, Fulton, and
4 Hanson 2015), especially through changes in growth rate (Brooks, Fulton, and Hanson
5 2015). Finally, even consumers can exhibit altered stoichiometry when exposed to
6 contaminants as seen in the larvae of the damselfly *Enallagma cyathigerum* (i.e., a food
7 resource for predators), which had higher N:P ratios when exposed to the pesticide
8 chlorpyrifos (Janssens, Op de Beeck, and Stoks 2016).

9 Elemental resource supply affects the production of secondary toxic metabolites
10 in plants and phytoplankton, which is further altered in the presence of anthropogenic
11 contaminant exposure. Several studies have investigated the stoichiometry of plant
12 secondary compounds, such as polyphenols (Bryant, Chapin III, and Klein 1983; Rivas-
13 Ubach et al. 2012; Nuttens et al. 2016). For example, Nuttens et al. (2016) showed that
14 higher N:P ratios in culture media led to lower concentrations of phenolic compounds in
15 the aquatic plant, *Myriophyllum spicatum*, and that this effect disappeared when plants
16 were exposed to the herbicide mesosulfuron. Elemental stoichiometry is also associated
17 with eukaryotic and prokaryotic toxin production (Van de Waal et al. (2009; 2014). As
18 many of these toxins are N-rich (i.e. low C:N), the concentration and supply ratios of
19 C:N, and N:P influence the production of these toxins, with high N supply compared to
20 demand increases N-rich toxin production (Van de Waal et al. (2009; 2014). In turn,
21 toxin production is sensitive to anthropogenic exposure. For example, *Microcystis*
22 populations produce more of the cyanotoxin, microcystin-LR, when exposed to the
23 antibiotic amoxicillin (Liu et al. 2014). These studies highlight the complex interactions

1 between elemental supply ratios, secondary metabolic toxin production physiology, and
2 their interaction with anthropogenic contaminant exposure.

3 Ecotoxicological investigations are often divided into two broad categories,
4 studies of contaminant exposure (bioconcentration, bioaccumulation, biomagnification)
5 and of contaminant effects (nonlethal and lethal), each of which can be influenced by
6 stoichiometric constraints. The effects of elemental imbalances on metabolism could also
7 alter the uptake (Figure 1, arrow 3), biotransformation, and elimination of contaminants
8 (Figure 1, arrow 4) with implications for contaminant fate and toxicity. For example, the
9 uptake of silver nanoparticles by algae, through an unknown mechanism, was higher
10 under P-rich compared to P-poor conditions (Conine and Frost 2017). In this case, P-rich
11 algae significantly reduced the toxicant concentration in water that, in turn, altered the
12 toxic impacts on animal consumers. Contaminants may also alter the uptake of elements
13 through modification of feeding behavior or digestive and assimilative processes (e.g.,
14 Frost and Tuchman 2005; Fulton et al. 2010). Several examples of this have been
15 explored for *Daphnia*, which alters its feeding rates in response to P availability
16 (Darchambeau, Faerøvig, and Hessen 2003; Karimi et al. 2007) and therefore may
17 experience increased exposure to food-borne toxicants (Karimi et al. 2007).

18 Stoichiometric constraints might also have indirect impacts on the bioaccumulation of
19 contaminants by organisms. The bioaccumulation/bioconcentration of lipophilic toxicants
20 can be affected by N or P-limitation that cause the accumulation of lipid stores both
21 producers (Kilham 1998) and consumers (Wagner et al. 2014). Indirect effects have been
22 investigated using both theoretical models (Peace, Poteat, and Wang 2016) and
23 experimental approaches (Wang and Dei 2006; Karimi et al. 2007; Arce-Funck et al.

1 2018). Those studies concluded that well-balanced elemental resources increase
2 organismal growth and dilute contaminants due to increased biomass.

3 Food quality and contaminants interact to affect the survival, growth, and
4 reproduction of producers and consumers. For example, *Daphnia* feeding on a low P diet
5 had a higher LC50 (lethal concentration required to kill 50% of the population) compared
6 to those feeding on P-rich diets when exposed to fluoxetine (Hansen et al. (2008). While
7 it is unknown what mechanism provides this decrease in toxicity, it has also been
8 displayed with *Daphnia* and other toxicants (e.g., glyphosate; (Lessard and Frost 2012)).
9 However, elemental limitation does not always confer protection against toxicity in
10 *Daphnia*. For example, elemental limitation increases toxicity when *Daphnia* were
11 exposed to imidacloprid (Ieromina et al. 2014) and silver nanoparticles (Conine and Frost
12 2017). This food quality contaminant interaction is not specific to planktonic consumers
13 and has also been found in a detritivore invertebrate, *Gammarus fossarum*. A higher
14 tolerance occurred in *G. fossarum* consuming high P diets after an acute exposure to
15 dissolved silver (Arce-Funck et al. 2016). While feeding on high P-resources during a
16 chronic, environmentally realistic exposure to cadmium tended to synergistically increase
17 the impact of the contaminant (Arce-Funck et al. 2018).

18 These examples highlight the need for controlled studies that examine each of the
19 many factors (e.g., timing, nutrient, and contaminant concentration gradients, exposure
20 duration) that may be important in understanding toxicity in a stoichiometric context.
21 However, these examples also suggest that organismal responses to both stoichiometric
22 and toxic stressors largely depend on the nature of the toxic compounds investigated,
23 duration of exposure, and the organism's biology. Coupled to these controlled studies, we

1 see the importance of future studies aimed at understanding environmental gradients of
2 chemical contaminants and complexity, particularly those substances with biological
3 activities mechanistically capable of altering molecular targets (e.g., transporters, efflux
4 pumps) and ecological functions in aquatic ecosystems (Rosi-Marshall and Royer 2012).
5 Similarly, future studies should consider environmental gradients of elements, including
6 conditions of elemental limitation and excess, that can alter growth rates, growth
7 efficiencies, and the ability of organisms to withstand changes in maintenance
8 requirements and other toxicant-induced stressors.

9

10 **Ecotoxicology and the stoichiometry of populations and communities.** Elemental
11 imbalances on producers and consumers have strong effects on their population dynamics
12 by altering population growth rates, maximum population size, and population
13 permanency. Such stoichiometric effects could modify adverse outcomes elicited by
14 contaminants at the population and community levels (Figure 1, arrow 6). For example,
15 populations of P-limited green algae are more sensitive to ibuprofen and less sensitive to
16 common high blood pressure and epilepsy medications compared to P-replete algae
17 (Grzesiuk, Wacker, and Spijkerman 2016). These indirect, elemental-driven differences
18 in toxicity could alter biomass production in lower trophic levels and food quality
19 available for predators, which add to the direct effects of the elements themselves.

20 Different species are also known to exhibit varying sensitivities to elemental
21 imbalances and/or contaminants, which will alter species composition in food webs and
22 community-level interactions (Figure 1, arrow 11). For example, antibiotic exposure of a
23 microbial community found on decomposing leaves resulted in a community shift in

1 favor of fungi over bacteria (Bundschuh et al. 2009). The antibiotic-exposed leaves with
2 more fungi were then preferentially consumed by invertebrate shredders, which suggests
3 interactive effects of elements and toxicants on the population and community dynamics
4 of heterotrophic food webs (Bundschuh et al. 2009). Changes in population size either
5 from poor nutrition associated with elemental imbalances or due to chemical toxicity may
6 also affect the total uptake and sequestration of contaminants with subsequent effects on
7 the exposure of other organisms (Figure 1, arrow 8). Despite their potential to occur,
8 community-level effects of stoichiometric toxicology are perhaps the least studied among
9 the different levels of biological organization.

10

11 **Ecosystems and stoichiometric ecotoxicology.** The field of ecological stoichiometry has
12 long studied how changes in the elemental nutrition of animals could affect the rate and
13 ratios of elemental cycling (Figure 1, arrow 5), which feedback into the supply of
14 elements available at the base of the food web (Figure 1, arrow 7; (Elser and Urabe
15 1999)). Similarly, toxic chemicals may alter metabolic states or the elemental
16 assimilation and excretion, with effects on the rate and ratios of elemental recycling
17 (Taylor et al. 2016). While this could be due to direct changes in organism metabolism, it
18 may also be mediated through changes to populations (Figure 1, arrow 9) or community
19 composition (Figure 1, arrow 10) as different life-stages and species may have distinct
20 stoichiometries (Bullejos et al. 2014).

21 The transfer of energy and matter across habitat boundaries, such as from aquatic
22 to terrestrial communities is well documented. Classic examples include aquatic insect
23 emergence being fed upon by riparian spiders (Kato et al 2003, Marczak and Richardson

1 2007) and birds (Murakami and Nakano 2002), or reciprocal transport from land to water
2 by terrestrial insects feeding trout (Nakano and Murakami 2001) and hippopotamus
3 transfer of savanna grassland nutrients to sub-Saharan African streams (Sabalusky et al.
4 2015). Given the breadth of knowledge linking nutrient availability to population and
5 community structure, as detailed above, it is apparent that C:N:P ratios of basal resources
6 likely influence organism and material movement across habitat boundaries.
7 Ecotoxicology can be linked to these subsidies as organically bound contaminants (e.g.,
8 PCBs, Walters et al. 2008, 2009) often accompany these movements of organisms. Yet,
9 to date, few studies have investigated questions related to these potential connections
10 (Arce-Funck et al., 2013; Fernandes et al., 2009) between biomass and composition of
11 organisms and toxins or toxicants, and the nutrient stoichiometry of food resources. There
12 is thus a need to better understand how stoichiometric mechanisms may alter the results
13 of ecotoxicological studies at the community and ecosystem levels.

14

15 **Modeling Advances in Stoichiometric Ecotoxicology**

16 Throughout its history, mathematical modeling has been a powerful tool with aims to
17 provide ways to organize ideas, develop hypotheses, and offer a qualitative understanding
18 of complex systems while making useful predictions. While traditional ecotoxicological
19 mechanistic modeling efforts take a variety of approaches to predict the effects of diverse
20 chemical contaminants on organismal responses (usually survival, growth, and
21 reproduction), many models assume, implicitly or explicitly, a nutrient-rich environment
22 where elemental limitation is absent. This assumption of a nutrient-replete environment
23 likely reflects the norm for empirical ecotoxicity studies, as noted above, where nutrient

1 media (in the case of plants, cyanobacteria, and algae) or food (for invertebrates and fish)
2 is sufficiently nutrient-rich to not overtly stress model organisms. In contrast, ES models
3 explicitly include elements to discover how the elemental composition of organisms
4 shapes their ecology, population dynamics, and ecosystem function. These models have
5 not generally considered contaminant effects on the causes and consequences of
6 elemental imbalances on organisms. Here we present brief overviews of models in these
7 two fields, consider recent modeling efforts that aim to integrate them, and discuss future
8 directions.

9

10 **Ecotoxicological Modeling.** The development of ecotoxicological models over the last
11 few decades has significantly contributed to interpreting how contaminants move and
12 accumulate throughout aquatic foods webs, as well as how they affect organisms (Hallam
13 and De Luna 1984; Mackay, Puig, and McCarty 1992; Gobas 1993; Ankley et al. 1995;
14 Wang, Fisher, and Luoma 1996; Kooijman and Bedaux 1996; Arnot and Gobas 2004;
15 Arnot and Gobas 2006; Pieters et al. 2006; Ashauer, Boxall, and Brown 2007; Wang et
16 al. 2008; Ashauer and Brown 2008; Bontje et al. 2009; Borgå et al. 2014; Huang, Wang,
17 and Lewis 2014, 2015; Huang and Wang 2016). These models take a variety of
18 approaches ranging from steady-state models of simplified systems with one to two
19 ordinary differential equations to more complex size-structured systems of partial
20 differential equations. While some of these models focus solely on contaminant exposure
21 (i.e. Arnot and Gobas 2006 balances uptake, assimilation, and elimination processes to
22 model bioaccumulation and Wang, Fisher, and Luoma 1996 uses kinetic models to
23 predict the bioaccumulation of trace elementals in mussels), many models simultaneously

1 model both contaminant exposure and effects (i.e. Huang, Wang, and Lewis 2015
2 dynamically model contaminant concentrations in organisms and how these body burdens
3 impact population dynamics). One of the best-tested modeling frameworks
4 systematically applied in ecotoxicology follows Dynamic Energy Budget (DEB) theory
5 and has been included in international risk-assessment guidance for its relevance in
6 analyzing ecotoxicity data (Jager, Gudmundsdottir, and Cedergreen 2014). While DEB
7 theory describes energy flows within an individual organism (Kooijman and Kooijman
8 2010; Nisbet et al. 2000), it aims to capture the quantitative aspects of metabolism and is
9 explicitly based on the principles of conservation of energy and mass (Sousa et al 2010).
10 The model can be applied and adapted to any organism with changes in species-specific
11 parameters. A recent Scientific Opinion for the European Food Safety Authority (EFSA)
12 noted the potential applicability of DEB models coupled to toxico-kinetic/toxicodynamic
13 effect models for regulatory risk assessment (Ockleford et al. 2018).

14 The “Add-my-pet” online database serves as a curated and constantly expanding
15 inventory of DEB parameters for thousands of species
16 (https://www.bio.vu.nl/thb/deb/deblab/add_my_pet/index.html, 2653 species, November
17 2020). The standard model describes how an organism ingests food, assimilating some of
18 the food as energy, and egesting the rest. Inside the organism, the assimilated energy is
19 stored in “reserve” components from which the organism allocates energy towards two
20 branches: growth and somatic maintenance and maturation/reproduction and maturity
21 maintenance. The independence of the reserve dynamics of food availability is a key
22 aspect of DEB theory and provides the individual with some protection against
23 environmental fluctuations and some control over its own metabolism (Sousa et al. 2010).

1 Contaminant effects are integrated into DEB models (nicknamed “DEBtox”, see
2 Jager and Zimmer (2012) and Jager (2016)) either by allowing a toxicant to impact
3 (change the value of) DEB parameters (e.g. (Jager et al. 2014)) or through the addition of
4 sub-models that connect the DEB model to toxicant processes (Klanjscek et al. 2012;
5 2013). Moreover, DEBtox has been used to investigate the effects of multiple stressors at
6 population and ecosystem scales. When stressors (e.g. food and contaminant) interact,
7 they can have different effects (depending on the mode of action) at higher levels
8 (individual, population, or ecosystem) compared to the stressor in isolation, as was
9 demonstrated for *Daphnia* models by (Martin et al. 2014). It is unknown *a priori* if the
10 effects will be additive, synergistic, or antagonistic (Galic et al. 2018). In Galic et al.
11 (2017), for example, the exposure to toxicants targeting maintenance, feeding, and
12 reproduction on a freshwater amphipod (*Gammarus pseudolimnaeus*) disproportionately
13 weakened consumer-resource interactions under warming at the ecosystem level but had
14 no effect at the population level.

15 While these ecotoxicology modeling efforts have enhanced understandings and
16 improved predictions of the adverse effects of contaminants, they often neglect to
17 incorporate stoichiometric constraints. On the other hand, the following section describes
18 stoichiometric modeling approaches, which have largely been developed without
19 considering contaminant stressors.

20

21 **Ecological Stoichiometric Modeling.** An approach often used to study the effects of
22 elemental imbalances on plants, animals, and microbes involve tracking the intake,
23 acquisition, and release of multiple elements simultaneously. These models explicitly

1 incorporate a mass balance perspective insomuch that elements acquired in excess of
2 growth, reproduction, or maintenance will be released (Frost, Xenopoulos, and Larson
3 2004; Frost and Tuchman 2005). Following this, element-specific uptake or consumption
4 can be coupled to elemental use for new biomass or maintenance to yield estimates of an
5 ideal resource mixture that maximizes growth or reproduction (Sterner 1997; Frost and
6 Elser 2002; Anderson and Hessen 2005). By altering the elemental mixture available for
7 acquisition, these models yield predictions about the strength and direction of elemental
8 constraints on organismal performance including slower growth, reduced reproduction, or
9 alterations in nutrient release.

10 Elemental imbalances and their constraints across trophic levels have been
11 incorporated into dynamical system models of population as well. Andersen (1997) took
12 the classical Rosenzweig MacArthur variation of the Lotka-Volterra equations and
13 incorporated nutrient-deficient growth by modifying the density dependence of the
14 producer's growth rate and the grazer's growth efficiency. Following Andersen's
15 approach, Loladze, Kuang, and Elser (2000) formulated a producer-grazer model (LKE
16 model) of the first two trophic levels of an aquatic food chain (algae-*Daphnia*) that
17 incorporated the assumption that both producers and grazers are chemically
18 heterogeneous organisms composed of two essential elements, C and P. The model
19 allows the P:C of the producer to vary above a minimum value but restricted the P:C
20 variation in grazer biomass. Introduction of these stoichiometric constraints significantly
21 affects the population dynamics and stability properties of the system, such as exhibition
22 of the paradox of "energy enrichment" empirically supported by Urabe et al. (2002).

1 The growing library of models that have extended the LKE model under the
2 theory of ES is mechanistic and robust. These models have incorporated dynamical
3 consequences of elemental limitation (Loladze, Kuang, and Elser 2000) and effects of
4 elements in excess (Peace et al. 2013). They have also been expanded to incorporate
5 multiple elements (Grover 2002; 2003; 2004) and two competing species (Loladze et al.
6 2004; Xie, Fan, and Zhao 2010). They can explicitly track free elements in the
7 environment (Wang, Kuang, and Loladze 2008; Peace, Wang, and Kuang 2014), and
8 determine important trophic transfer efficiencies (Peace 2015). The ES modeling
9 framework has been expanded to include spatial dynamics (Rana et al. 2018), incorporate
10 multi-stage dynamic processes (Jiang et al. 2019), and capture evolutionary dynamics
11 (Yamamichi et al. 2015). These stoichiometric models incorporate the effects of both
12 food quantity and food quality into a single framework that produces rich dynamics (i.e.
13 stabilizing predator-prey systems) and allow more than one predator species to coexists
14 on a single prey, producing multiple basins of attraction and bi-stability, exhibiting
15 multiple types of bifurcations, and allowing evolutionary rescue. This ES modeling
16 framework applies equally to phenomena at sub-organismal levels as well as the whole
17 biosphere level (Sternner and Elser 2002). Despite these advances, models developed
18 under the ES framework typically disregard the effects of contaminants. The above
19 briefly summaries highlight the fact that the majority of modeling efforts in
20 ecotoxicology and ES have largely been developed separately. Below we discuss the
21 opportunities for theoretical approaches to better integrate these two fields within single
22 modeling frameworks.

23

1 **Stoichiometric Ecotoxicological Modeling.** While there are modeling efforts that
2 consider contaminant stressors along with element limitation (i.e. Bontje et al. (2009) and
3 Ankley et al. (1995)), these models are not explicitly stoichiometric and do not allow for
4 multiple elemental constraints where the element limiting growth can change with
5 environmental element and light availabilities. Here, we propose integrative modeling
6 frameworks with promising potential to explicitly incorporate elements and contaminants
7 as well as integrative effects between them.

8 Existing theoretical frameworks in ES and ecotoxicology actually have similar
9 structures but they utilize different perspectives. For example, ES uses an element
10 perspective to develop mass-balance models of threshold elemental ratios (i.e. Frost,
11 Xenopoulos, and Larson 2004, Frost and Tuchman 2005) and ecotoxicology uses a
12 contaminant perspective to develop mass-balance models of bioaccumulation (i.e. Gobas
13 1993). While the mass-balance modeling approaches here are similar, they employ
14 different assumptions on parameters as the biological regulation of elements can
15 significantly differ from regulation of contaminants. Both fields use dynamical
16 population models, where ES uses a nutrient perspective to track essential elements
17 throughout food webs and their influence of population dynamics (i.e. Loladze, Kuang,
18 and Elser 2000) and Ecotoxicology uses a contaminant perspective to track the fate of
19 contaminants throughout food webs and their effects on population dynamics (i.e. Huang,
20 Wang, and Lewis 2014, 2015). These modeling approaches can be unified by integrating
21 contaminants into the ES models and/or by integrating nutrients into the ecotoxicological
22 model as shown in Figure 2.

1 DEB theory offers a modeling framework with the potential to analyze
2 ecotoxicological issues in which stoichiometry plays an important role. This would
3 require a more detailed or explicit stoichiometric framework than is currently
4 incorporated in DEBtox studies. The standard DEB model is the simplest, non-
5 degenerated, model among a family of models (Marques et al. 2018) that describes the
6 full life cycle of an animal feeding on a single food source with constant elemental
7 composition. Thus, this model does not generally incorporate or examine stoichiometric
8 mechanisms per se. However, multiple-reserve (instead of single-reserve) DEB models
9 have been developed to consider stoichiometry explicitly. Kooijman (1998) proposed a
10 generalizable model of two reserves, specifically citing applications to element-limited
11 algal growth, such as the classic Droop models (Droop 1974). Sperfeld et al. (2017)
12 described a DEB model with separate nutrient reserves (C and N) coupled with metabolic
13 pathways that may be impacted by changes in C or N mass balance. DEB models that
14 consider stoichiometric constraints have been developed for a wide variety of
15 applications, ranging from tracing isotope flows (Pecquerie et al. 2010) to describing the
16 symbiosis between coral and photosynthetic algae (Muller et al. 2009; Cuning et al.
17 2017) to analyzing stoichiometric constraints on population dynamics (Kuijper et al.
18 2004a; 2004b; Kooijman 1998; Kooijman, Andersen, and Kooi 2004). However, to our
19 knowledge, no study has utilized a stoichiometrically-explicit DEB model to explain or
20 predict ecotoxicological responses or adverse outcomes.

21 ES offers a complementary modeling framework to investigate the influence of
22 elemental imbalances on responses of organisms to contaminants while simultaneously
23 considering the effects of contaminants on ecosystem processes (Danger and Maunoury-

1 Danger 2013). Toxicity could be incorporated into ES models by altering functions of
2 organismal nutrient intake and/or by changing element-metabolism relationships with
3 chemical exposure. For example, Frost and Tuchman (2005) used this type of model to
4 demonstrate how reduced growth and increased element release rates from two benthic
5 invertebrates resulted from lower assimilation efficiencies in animals consuming nutrient-
6 poor leaf material grown under elevated CO₂ concentrations. Peace, Poteat, and Wang
7 (2016) and Hassan et al. (2018) developed models that incorporated the effects of
8 concurrent element and toxicant stressors on population dynamics and the trophic transfer
9 of toxicants. These models dynamically model the bioaccumulation of toxicants under
10 varying nutrient resources, capturing stoichiometric contaminant exposure dynamics, and
11 incorporate the effects of toxicant and nutrient stressors into growth dynamics. In these
12 models, elemental changes in the environment impact organismal growth rates and thus
13 population dynamics, which can drive changes in contaminant concentrations within the
14 organisms. Additionally, they captured and explored a phenomenon called the Somatic
15 Growth Dilution (SGD) effect, observed empirically (Karimi et al. 2007), where
16 organisms experience a greater than proportional gain in biomass relative to toxicant
17 concentrations when consuming food with high nutritional content vs. low quality food.
18 In addition to influencing life-history traits, specific contaminants can impact organism
19 elemental composition, (Ni, Kelly, and Arhonditsis 2017; Baudrot et al. 2018; Danger
20 and Maunoury-Danger 2013). Future model iterations should consider the impact of
21 contaminants on organism elemental compositions. While there have been few
22 toxicological applications, ES models appear to have considerable potential to

1 significantly improve our understanding of the effects that chemical contaminants have
2 on organisms and ecosystems (Hansen et al. 2008).

3

4 **Areas of future research**

5 Organisms, populations, communities, and ecosystems all function in a world of
6 contaminant and nutrient mixtures. We are now at a point where we can aim for a more
7 mechanistic understanding of the combined effects of contaminants and elements at each
8 level of biological organization. To achieve meaningful advances, some of the
9 fundamental discoveries and models developed in the fields of ES and ecotoxicology
10 should be considered and, if possible, merged into a unified framework, as proposed in
11 Figure 2. These considerations could include: 1) determining nonlinear responses to
12 elements *and* contaminants and their interactions, 2) potential feedbacks between the
13 effects of elements and contaminants at different scales, and 3) integrating molecular
14 responses to better link to and understand the underlying associated biological changes.
15 The first steps would help define an integrated conceptual framework and inform specific
16 research goals to advance our understanding of the effects of contaminant-nutrient
17 combinations that more closely reflect natural environments.

18 Nonlinearities in the effects of individual elements and contaminants could alter
19 our predictions of interactions between these substances. At the organismal level,
20 nonlinear physiological responses to individual contaminants have been well documented
21 for decades. A classic example is the hormesis response in which organisms exhibit a
22 positive response to a contaminant at low doses and a negative, deleterious response at
23 higher levels (Calabrese and Baldwin 2002). More recent work suggests biological

1 adaptation as a potential mechanistic explanation for the otherwise phenomenological
2 hormetic response (Calabrese and Mattson 2017). Another landmark study examined and
3 defined nonmonotonic responses to a suite of endocrine disrupting contaminants
4 (Vandenberg et al. 2012). Classic models highlight nonlinearities in physiological
5 responses to essential and nonessential metals (Karimi and Folt 2006). Recent work also
6 suggests that nonlinearities are not limited to toxicants and can also occur in response to
7 variable element (e.g., P and N) regimes. Specifically, high inputs of N and P can cause
8 negative effects in forms that are not otherwise explained by other nutrient deficiencies
9 and that would not be expected based on ES theory (Costello et al. 2018). Additional
10 studies are needed to understand the negative effects of high element doses, particularly
11 as global element supplies continue to increase, and eutrophication of diverse ecosystems
12 become more evident (Howarth 2008). Overall, nonlinear responses are neither rare, nor
13 limited to certain types of substances, particularly when substances interact with multiple
14 biological targets, and therefore have high potential to influence the combined effects of
15 multiple nutrients and contaminants. Thus, there is a need to develop predictive models
16 that examine and explain the influence of nonlinear responses on the combined and
17 interactive effects of multiple substances.

18 Nonlinear interactions and dynamics illustrate how complexities in the behavior
19 and effects of elements and toxicants pose a challenge for how we distinguish among
20 different types of substances. In some cases, substances considered to be elements
21 function more like contaminants (Karimi and Folt 2006; Stumm and Morgan 2012).
22 Several different classification systems exist that attempt to make clear, useful
23 distinctions among types of substances based on different characteristics. For

1 convenience in this paper, we have used perhaps the most simplistic classification system
2 comprised only of nutrients versus contaminants. However, developing a unified
3 framework that combines knowledge from the ES and ecotoxicology fields requires us to
4 revisit common classification systems in terms of how they may help us understand the
5 behavior and effects of elements and toxicants in mixtures. This would help us organize
6 and predict potentially complex, higher-order interactions among substances. Part of this
7 could be mediated through a better understanding of the molecular components involved
8 in the processes underlying element and toxicant effects as described by the various DEB
9 processes. Well-known examples of such systems include: 1) classes of metals (e.g.,
10 Class A, B), 2) macro-elements, essential trace metals, nonessential trace metals (Karimi
11 and Folt 2006; Karimi 2010), 3) organic contaminant functions based on K_{ow} values, and
12 4) contaminant functions based on specific molecular interactions. Moving toward a
13 stoichiometric ecotoxicology framework requires us to define substances in ways that
14 help us understand whether there are emergent patterns in how different types of
15 substances interact and are organized at different scales.

16 A recent development in ecotoxicology to conceptualize “big-picture” impacts of
17 contaminants has been made through the adverse outcome pathway (AOP) framework
18 (Ankley et al. 2010). This framework could offer a way to classify substances in the
19 context of stoichiometric ecotoxicology and connect biological scales. AOPs have been
20 developed to conceptualize the knowledge linking a molecular initiating event of a
21 contaminant to an adverse outcome at a relevant level of biological organization via key
22 events (KE) and key event relationships. They can be used to link molecular responses to
23 underlying biological changes, an area of future work identified in figure 2. KEs can

1 occur at any level of biological organization and can be connected to DEB processes
2 (Murphy et al. 2018a; Murphy et al. 2018b). Traditional AOPs are chemically agnostic
3 and encode causal, not quantitative, relationships. However, the development of
4 quantitative AOPs (qAOPs) is an emerging field (Perkins et al., 2019). In qAOPs,
5 quantitative key event relationships, such as dose-response and response-response
6 relationships, can be influenced by ecotoxicological context through modulating factors
7 (Conolly et al. 2017). Modulating factors alter quantitative key event relationship
8 between two KEs, such as to change the shape of a response-response relationship.
9 Examples of modulating factors include the impact of environmental, genetic, disease, or
10 nutritional variations on the KE relationship (Villeneuve et al. 2014). Informing
11 appropriate modulating factors by nutrient-chemical interactions and relationships studied
12 in stoichiometric ecotoxicology will serve to refine quantitative relationships in qAOPs.
13 Connecting the established frameworks of DEB modeling and AOPs in circumstances
14 where elements might behave as stressors (under nutrient limitation or in excess) or
15 where contaminants behave as nutrients (e.g., hormesis at low doses) could improve the
16 classification, down-stream modeling, and prediction of emergent properties.

17 Future work is also needed to examine potential feedbacks among nutrient and
18 contaminant exposures at different levels of biological organization. For example,
19 contaminant exposure at the organismal and population level may influence element
20 cycling at the ecosystem level by changing organism physiological processes (e.g.,
21 growth and excretion). In turn, element cycling may also impact contaminant toxicity,
22 through somatic growth dilution (SGD) and reductions in bioaccumulation (Karimi et al.
23 2007). To date, very little is known about the extent, strength, and type of such feedbacks

1 and the circumstances under which they may occur. Such questions are particularly
2 important to address to understand changes in contaminant and element inputs to
3 ecosystems, including increases in nutrient pollution to aquatic systems.

4 There is also a need to examine the combined effects of element and contaminant
5 exposure on organisms across age, size, or life-stages. Mathematical models have been
6 used to incorporate stage structure into population dynamics. Structured population
7 models are able to incorporate additional complexity related to an additional variable,
8 such as age, size, or life-stage. These come in the form of discrete matrix equations
9 (Leslie, 1947), systems of ordinary differential equations, or continuous partial
10 differential equations (Sinko and Streifer 1967). Within the context of ecotoxicology,
11 systems of differential equations may include multiple compartments representing
12 various stages of the life cycle and, correspondingly, different effects of toxicants on the
13 stages of the life cycle. These models can include density-dependent effects of
14 populations. It is straightforward to incorporate ecotoxicological effects for the various
15 life-stages in a matrix model (Emlen and Springman 2007; Erickson et al. 2014) or a
16 system dynamics model (Weller et al. 2014). Furthermore, these models can easily be
17 extended to model nutritional stress (Weller et al. 2014; Hanson and Stark 2011; Caswell
18 2006).

19 Continuously-structured population models can be useful as they are more
20 computationally stable when estimating parameters that are dependent on time or the
21 structured variable. There are many continuously structured population models that
22 incorporate food availability (Ananthasubramaniam et al. 2011), and some that implicitly
23 consider food quality (Nisbet and Gurney 1983). However, explicit modeling of

1 stoichiometric constraints remains relatively unexplored within this framework.
2 Ecotoxicological effects have also largely remained un-integrated with continuously-
3 structured population models. Recently, however, Huang and Wang (2016) proposed a
4 toxin-mediated size-structured population model allowing for different sensitivities to
5 toxicants depending on size, age, etc.

6
7 **Incorporating stoichiometric ecotoxicology into management decisions.** Advancing
8 stoichiometric ecotoxicology promises to reduce uncertainty during environmental risk
9 assessment and management activities. This may be particularly true for extrapolations
10 across laboratory to field scales of biological organization, which remain among the
11 biggest research needs to achieve a more sustainable environment (Furley et al. 2018;
12 Van den Brink et al. 2018). Brooks, Fulton, and Hanson (2015) recommended accounting
13 for stoichiometric influences on aquatic ecotoxicology studies with plants and algae by
14 balancing experimental pragmatism with environmental realism. Traditional
15 ecotoxicology assays were developed to ensure sufficient growth of primary producers,
16 which represents a common endpoint to understand adverse outcomes of contaminant
17 exposures. To predict real-world outcomes, future research on ecotoxicology needs to be
18 more representative of field-relevant elemental gradients. If nutrient assimilation and
19 elemental ratios of lower trophic levels are modified, those changes will cascade into
20 changes in interspecific competition, the efficiency of secondary production, and
21 community composition.

22 We also propose better development and use of knowledge about how
23 stoichiometric conditions influence the production of natural toxins by aquatic and
24 terrestrial organisms. This is of particular relevance to ecological risk assessments in

1 inland and coastal waters, where harmful algal blooms (HAB) can elicit profound
2 impacts on aquatic life and public health and often surpass anthropogenic chemical risks.
3 In fact, HABs are now recognized as transformative threats to water quality and
4 confounders of environmental assessment and management efforts, which often rely on
5 risk-based regulatory frameworks for the protection of public health and the environment
6 (Brooks et al. 2016). Whereas ecological studies and monitoring activities have
7 previously examined toxin concentrations, these efforts are routinely limited by the
8 absence of robust analytical quantitation of diverse toxins produced by specific HAB
9 species. This represents a critical consideration for water resource management because
10 HAB forming species are assumed to be producing toxins even if measured toxins are
11 below detection limits. These conservative measures decrease the exposure of toxins but
12 may come at an unnecessary cost. While water quality models can predict the occurrence,
13 intensity, and severity of HABs, by not incorporating toxin production these models
14 cannot predict the risks to human or ecosystem health. Developing predictive growth,
15 toxin production, and comparative toxicity models across stoichiometric nutrient
16 gradients are imperative for forecasting, diagnosing, and preventing ecological and
17 human health risks presented by algal toxins.

18

19 **Concluding Remarks**

20 Here we refer to the integration of ecotoxicology and ES theory as stoichiometric
21 ecotoxicology. The conceptual framework for stoichiometric ecotoxicology that we have
22 developed highlights the interactive impacts of elemental imbalances and chemical
23 contaminants within and across scales from sub-organismal to ecosystems. This
24 conceptual framework was built from existing empirical and theoretical examples that

1 link stoichiometry and toxicology. While an increasing amount of stoichiometrically
2 explicit models have provided insight into ecological interactions, there is much to be
3 gained by incorporating concurrent nutrient and toxicant stressors into models formulated
4 under the presented stoichiometric ecotoxicology framework. This unifying framework
5 offers the potential to deepen mechanistic understandings of the adverse outcomes of
6 chemicals across ecological scales and improve the predictive powers of
7 ecotoxicology.

8

9

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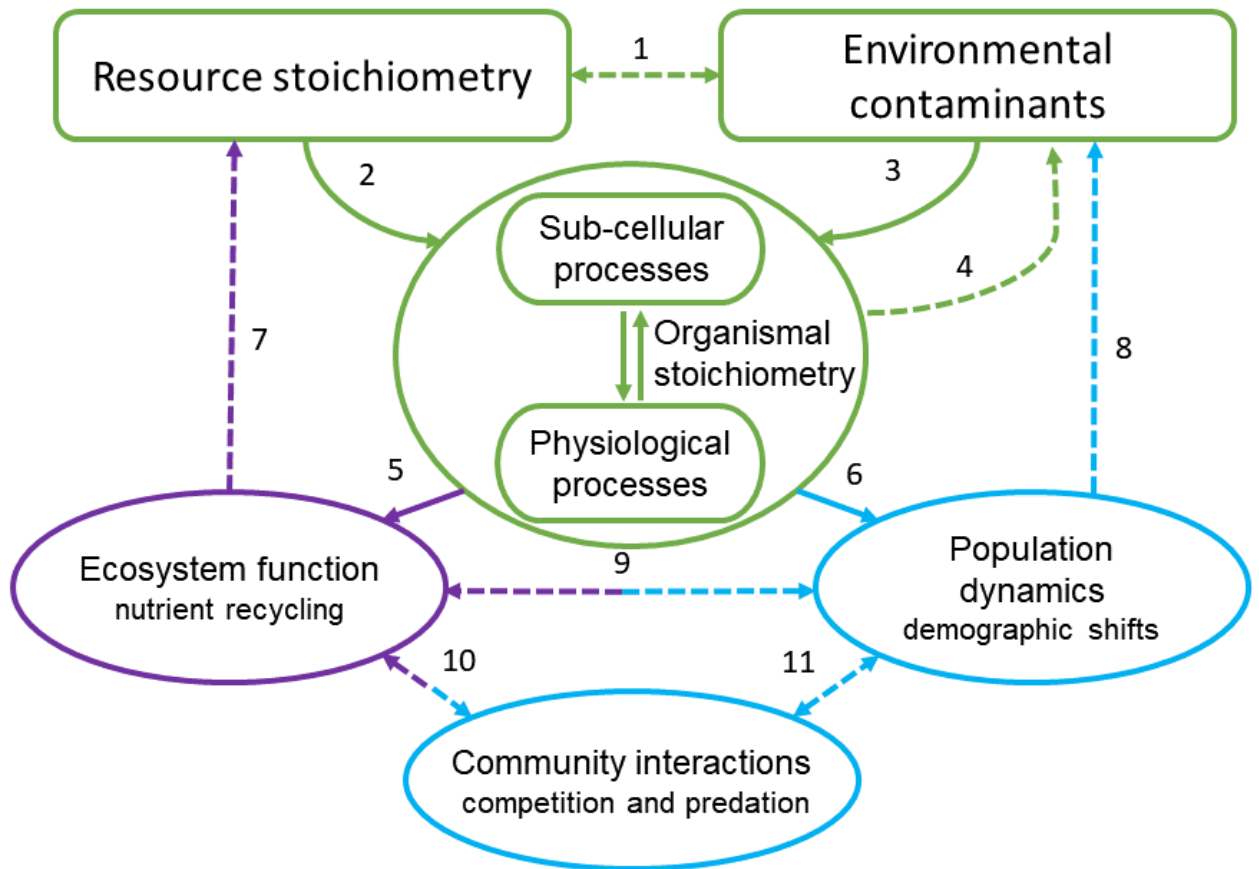
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STOICHIOMETRIC ECOTOXICOLOGY



1

Figure 1. Possible interactions of resource stoichiometry (i.e., nutrient ratios in the environment) and environmental contaminants on relationships from the sub-organismal level through ecosystem level. These effects include feedbacks within and between levels of organization. Colors represent three categories of interaction: Organismal and sub-organismal (green), population and community (blue), and ecosystem level stoichiometric ecotoxicology (purple). Dashed arrows represent indirect mechanisms and/or indirect feedbacks among components. Arrows are numbered and explained in the accompanying text.

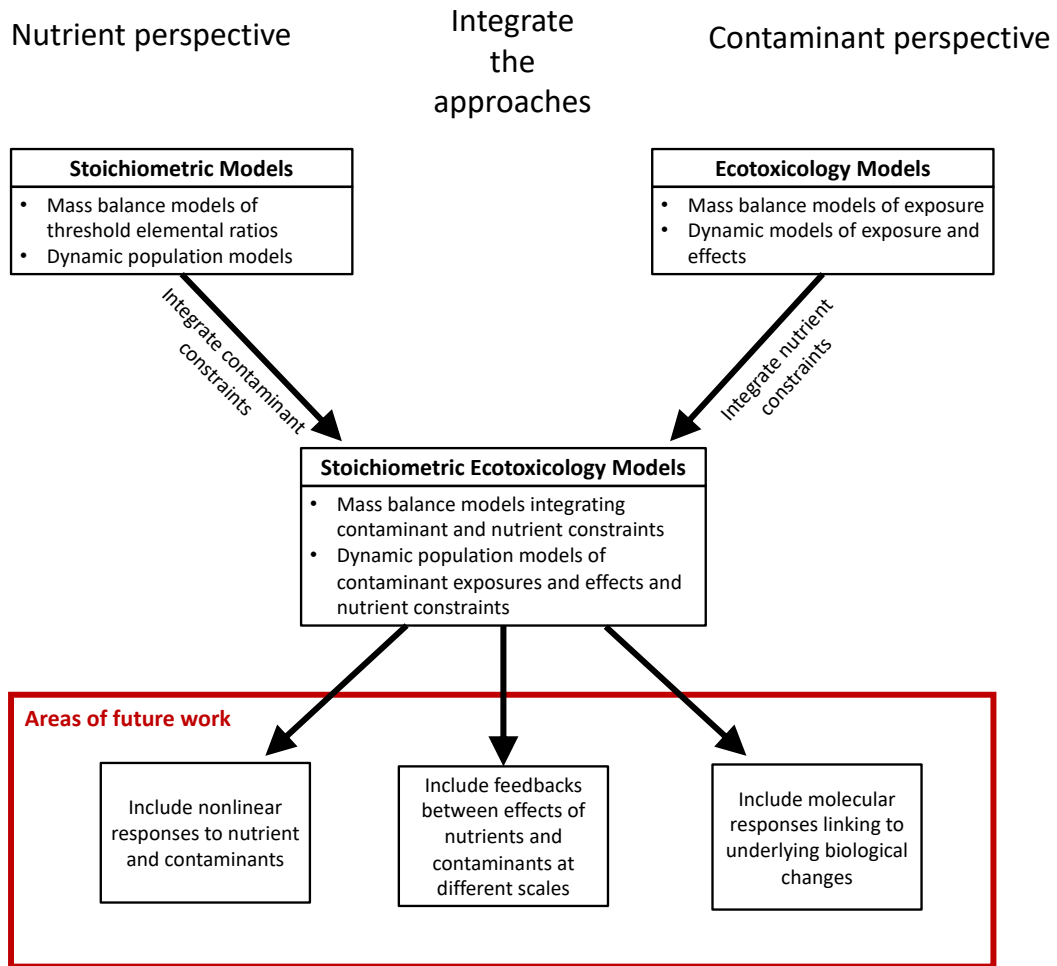
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2 **Figure 2.** Proposed integrative modeling frameworks have promising potential to
3 explicitly incorporate nutrients and contaminants, as well as, integrative effects between
4 them. Stoichiometric Ecotoxicology models can then be extended to incorporate and
5 guide areas of future work listed in the red box.
6

Table 1. Examples of ecotoxicology studies that have examined stoichiometric mechanisms.

Effect	Contaminant	Biological model / ecological process	Stoichiometric parameter	Study scale	Type of ecosystem	Type of approach	Reference
<i>Effects of the contaminant on dissolved nutrient availability and stoichiometry</i>							
	Aluminium	n.a.	Mineral nutrient stoichiometry	Ecosystem	Freshwater	Observational	Kopacek et al. 2000
	Aluminium	Microbial decomposers	Mineral nutrient stoichiometry	Community	Freshwater	Microcosms	Clivot et al. 2014
<i>Effects of basal resources stoichiometry on contaminant bioavailability in the medium</i>							
	Silver nanoparticles	Zooplankton	Mineral nutrient stoichiometry	Individual	Freshwater	Microcosms	Conine & Frost 2017
<i>Effects of the contaminant on basal resources C:N:P ratios)</i>							
	Triclosan	Plant	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	Fulton et al. 2010
	Atrazine	Periphyton	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	(Murdock and Wetzel 2012)
	Silver	Microbial decomposers	Plant detritus C:N:P ratios	Community	Freshwater	Microcosms	Arce Funck et al. 2013
	Silver nanoparticles	Phytoplankton	Primary producers C:N:P ratios	Community	Freshwater	Microcosms	Das et al. 2014.
	Titanium nanoparticles	Cyanobacteria	Primary producers C:N:P ratios	Community	Freshwater	Microcosms	Cherchi et al. 2015
	Isoproturon, Mesosulfuron	Plant	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	Nuttens et al. 2016
	Chlorpyrifos	Insect	Animal C:N:P ratios	Individual	Freshwater	Microcosms	Janssens et al. 2017
<i>Effects of resources elemental composition on consumers tolerance to contaminants</i>							
	Fluoxetine	Zooplankton	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	Hansen et al. 2008
	Triclosan	Plant	Mineral nutrient stoichiometry	Individual	Freshwater	Microcosms	Fulton et al. 2010.
	Glyphosate	Zooplankton	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	Lessard & Frost 2012
	Uranium, Arsenic	Plant	Mineral nutrient stoichiometry	Individual	Freshwater	Microcosms	Mkandawire & Dudel 2012
	Silver	Plant	Mineral nutrient stoichiometry	Individual	Freshwater	Microcosms	Bian et al. 2013
	Imidachloprid	Zooplankton	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	Ieromina et al. 2014
	Metolachlor	Plant	Mineral nutrient stoichiometry	Individual	Freshwater	Microcosms	Brooks et al. 2015
	Atrazine	Phytoplankton	Mineral nutrient stoichiometry	Population	Freshwater	Microcosms	Baxter et al. 2016
	Silver	Macroinvertebrate	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	Arce Funck et al. 2016
	Silver nanoparticles	Zooplankton	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	Conine & Frost 2017
	Cadmium	Macroinvertebrate	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	Arce Funck et al. 2018

Effects of resources stoichiometry on consumers bioaccumulation of the contaminant

Cadmium, Copper	Phytoplankton	Mineral nutrient stoichiometry	Population	Freshwater	Microcosms	Wang & Dei 2006
Mercury	Zooplankton	Primary producers C:N:P ratios	Individual	Freshwater	Microcosms	Karimi et al. 2007
n.a.	n.a.	Primary producers C:N:P ratios	n.a.	Freshwater	Theoretical model	Peace et al. 2016
Cadmium	Macroinvertebrate	Plant detritus C:N:P ratios	Individual	Freshwater	Microcosms	Arce Funck et al. 2018

Towards the use of stoichiometric approaches including non essential, potentially toxic elements

Diverse metals	Macroinvertebrates	Primary producers Multielemental ratios		Freshwater	Observational	Karimi & Folt 2006
Cadmium, Copper	Phytoplankton	Mineral nutrient stoichiometry	Population	Freshwater	Microcosms	Wang & Dei 2006

Interactive effects of contaminants and stoichiometry on ecosystem functions/processes

Zinc	Leaf litter microbial decomposition	Mineral nutrient stoichiometry	Community	Freshwater	Microcosms	Fernandes et al. 2009
Silver	Leaf litter microbial decomposition	Mineral nutrient stoichiometry	Community	Freshwater	Microcosms	Arce Funck et al. 2013
Atrazine	Periphyton primary production	Mineral nutrient stoichiometry	Community	Freshwater	microcosm	(Murdock, Shields Jr., & Lizotter Jr. 2013)
Silver nanoparticles	Phytoplankton primary production	Mineral nutrient stoichiometry	Community	Freshwater	Microcosms	Das et al. 2014
Silver	litter decomposition by macroinvertebrate	Plant detritus C:N:P ratios	Individual	Freshwater	Microcosms	Arce Funck et al. 2016
Cadmium	litter decomposition by macroinvertebrate	Plant detritus C:N:P ratios	Individual	Freshwater	Microcosms	Arce Funck et al. 2018

Effects of nutrient stoichiometry on the production of toxic compounds by organisms

Plant secondary compounds	Plant	Mineral nutrient stoichiometry	Individual	Terrestrial	Literature review	Bryant et al. 1983
Cyanotoxins	Cyanobacteria	Mineral nutrient stoichiometry	Population	Freshwater	Microcosms	Van de Waal et al. 2009
Plant secondary compounds	Plant	Mineral nutrient stoichiometry	Individual	Terrestrial	Field experiment	Rivas-Ubach et al. 2012
Cyanotoxins	Phytoplankton/Cyanobacteria	Mineral nutrient stoichiometry	n.a.	Freshwater	Literature review	Van de Waal et al. 2014
Cyanotoxins	Cyanobacteria	Mineral nutrient stoichiometry	Population	Freshwater	Microcosm	(Wanger et al. 2019)
Plant secondary compounds	Plant	Mineral nutrient stoichiometry	Individual	Freshwater	Microcosms	Nuttens et al. 2016