

1 **Disentangling food-web environment relationships: a review** 2 **with guidelines**

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29 is made available as Supplementary Information.

30 **Data sources:** The review is based upon the review of 463 papers published between
31 2006 and 2017. References are made available within the database.

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41 **Abstract**

42 Food webs represent energy fluxes and nutrient cycling, underpinning ecosystem
43 functioning. Whether and how interactions vary over environmental gradients is still
44 largely unknown. We reviewed the literature searching for systematic relationships
45 between structural food-web properties and environmental gradients. Temperature
46 and biotic factors are amongst the most addressed drivers on determining structural
47 food web properties. Most studies are local, replication is often lacking, and regional
48 generalities are difficult. The lack of a consistent theory predicting how food webs
49 change across environmental gradients, the diversity of objectives in food-web studies,
50 and the absence of a standardized methodology for studying them severely limit
51 progress in the field. Moving forward requires the establishment of a core set of
52 testable predictions, agreed standards for data collection and analysis, and the
53 development of geographically distributed experimental studies of food-webs
54 dynamics.

55 **Keywords:** biotic interactions; ecosystem services; environmental change; food webs.

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67 **1. Introduction**

68 The representation of communities as networks of species connected by trophic
69 relationships was first proposed by Charles Elton in the late 1920s (Elton, 1927).
70 Research on the topic increased during the '70s-'80s (Layman et al., 2015) and gained
71 new momentum recently, owing to the renewed interest in understanding
72 environmental change effects on biodiversity. Food webs provide a schematic
73 representation of energy processing and transformation while incorporating
74 information about species composition (nodes in network terminology) and the
75 distribution of trophic interactions (links connecting nodes) (Barnes et al., 2018). By
76 explicitly considering species, interactions, energy and matter fluxes, this framework
77 allows the exploration of the effects of environmental changes on different facets of
78 biodiversity.

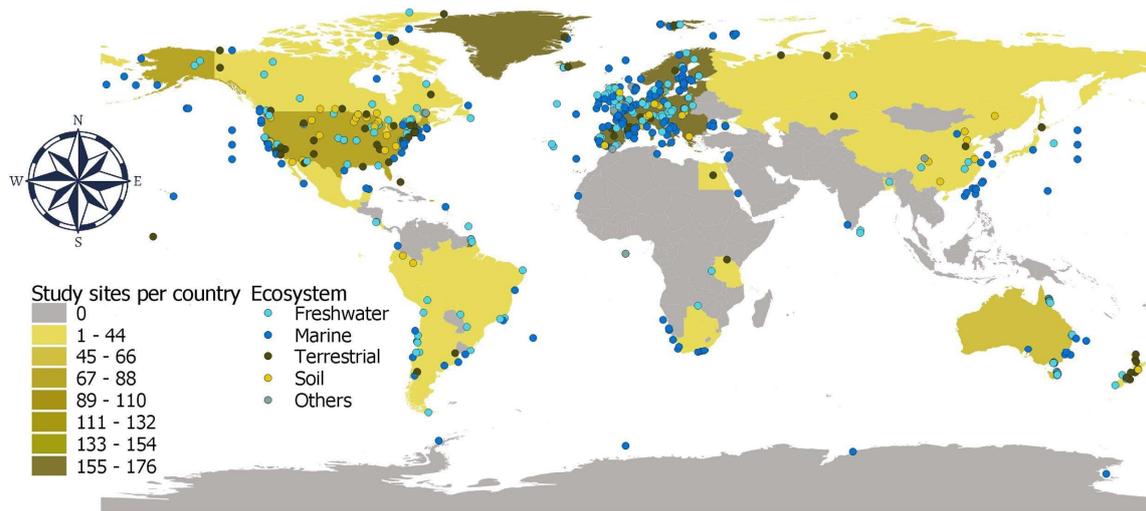
79 Species distributions and their interactions are driven by multiple environmental
80 characteristics, such as temperature and precipitation (Peterson et al., 2011; Post,
81 2013). Food webs provide the link between community ecology (and its building blocks,
82 individuals, species, and populations) and ecosystem ecology (flows of biomass, energy
83 and nutrients) (Thompson et al., 2012). If emerging functional food-web properties
84 such as trophic regulation, energy transfer efficiency, or primary and secondary
85 productivity, changed predictably along environmental gradients, then our ability to
86 anticipate the consequences of global environmental changes on ecological
87 communities would be considerably simplified.

88 Broad-scale biogeographical relationships such as the latitudinal diversity gradient,
89 species distribution-environment associations, species-area relationships, temperature-
90 body size rules, or species range-size frequency distributions, are often used to
91 anticipate some of the ecological consequences of global changes. It is less clear,
92 however, if aspects of food-web organization follow consistent changes along gradients
93 (Baiser et al., 2019), such as temperature and productivity (Mendoza & Araújo, 2019).

94 Previous reviews have addressed the effects of environmental gradients on ecological
95 networks (Pellissier et al., 2017; Tylianakis & Morris, 2017), concluding that these are
96 mainly related with changes on species composition, relative abundances, or

97 coevolutionary processes affecting interactions. Avenues for improvement in food web
 98 and network ecology have also been addressed (Cohen et al., 1993; Dunne, 2005; Ings
 99 et al., 2009; Thompson et al., 2012), recommending: 1) an accurate description of
 100 network nodes, 2) improved information on the links (reporting all links, preferentially
 101 quantitatively and based on observations), 3) defining the setting (spatial description,
 102 sampling temporal span and effort), 4) standardizing the data structure, 5) promoting
 103 collaborative efforts between researchers with different expertise, 6) strengthening the
 104 theoretical framework and mechanistic understanding (to improve predictability), 7)
 105 resorting to a more individual-based perspective, 8) integrating different types of
 106 ecological networks, 9) studying networks on natural gradients, and 10) encouraging
 107 manipulative experiments. These recommendations are intended to improve data
 108 quality, comparability and availability and drive the field towards a more predictive,
 109 mechanistic view.

110 We comprehensively reviewed the literature over the last decade (2006 to 2017),
 111 asking if structural food-web properties change consistently with environmental
 112 variation. We compiled a comprehensive database with 463 studies distributed
 113 worldwide (Figure 1 and Annex 1, Supporting Information).



114

115 Figure 1 – Study sites for the articles considered per major biome (Countries and EU + UK, Switzerland and Norway:
 116 light yellow to brown: number of studies per country).

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118 Our objectives were: i) search for the existence of relationships between environmental
 119 gradients and food-web structural properties, whether at the node, interaction or food
 120 web structural level; ii) assess the state-of-the-art of food web ecology and provide
 121 future perspectives.

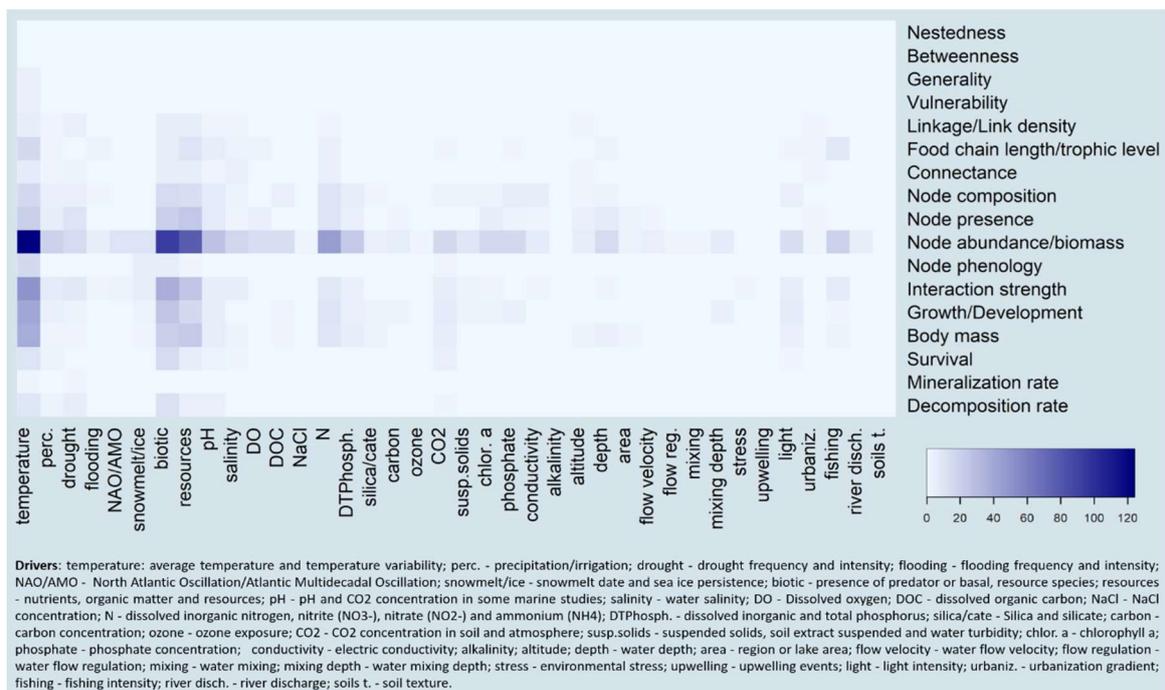
122 We analysed the number of articles showing significant relations between predictors
 123 and response variables and summarized this information in a database. The inclusion of
 124 articles in the database followed a two-step selection process: a) a search in the Web of
 125 Science with pre-determined keywords b) retaining those with at least three trophic
 126 interactors being related to an environmental gradient.

127

128 2. Environmental gradients and food webs

129 2.1. Effects on nodes and interactions

130 Two predictors stand out as showing significant relations with response variables
 131 (Figure 2): temperature and biotic-related variables (e.g. presence of predators). The
 132 response variable that was more frequently considered was node abundance/biomass
 133 (the abundance/biomass of a species, trophic or functional group).



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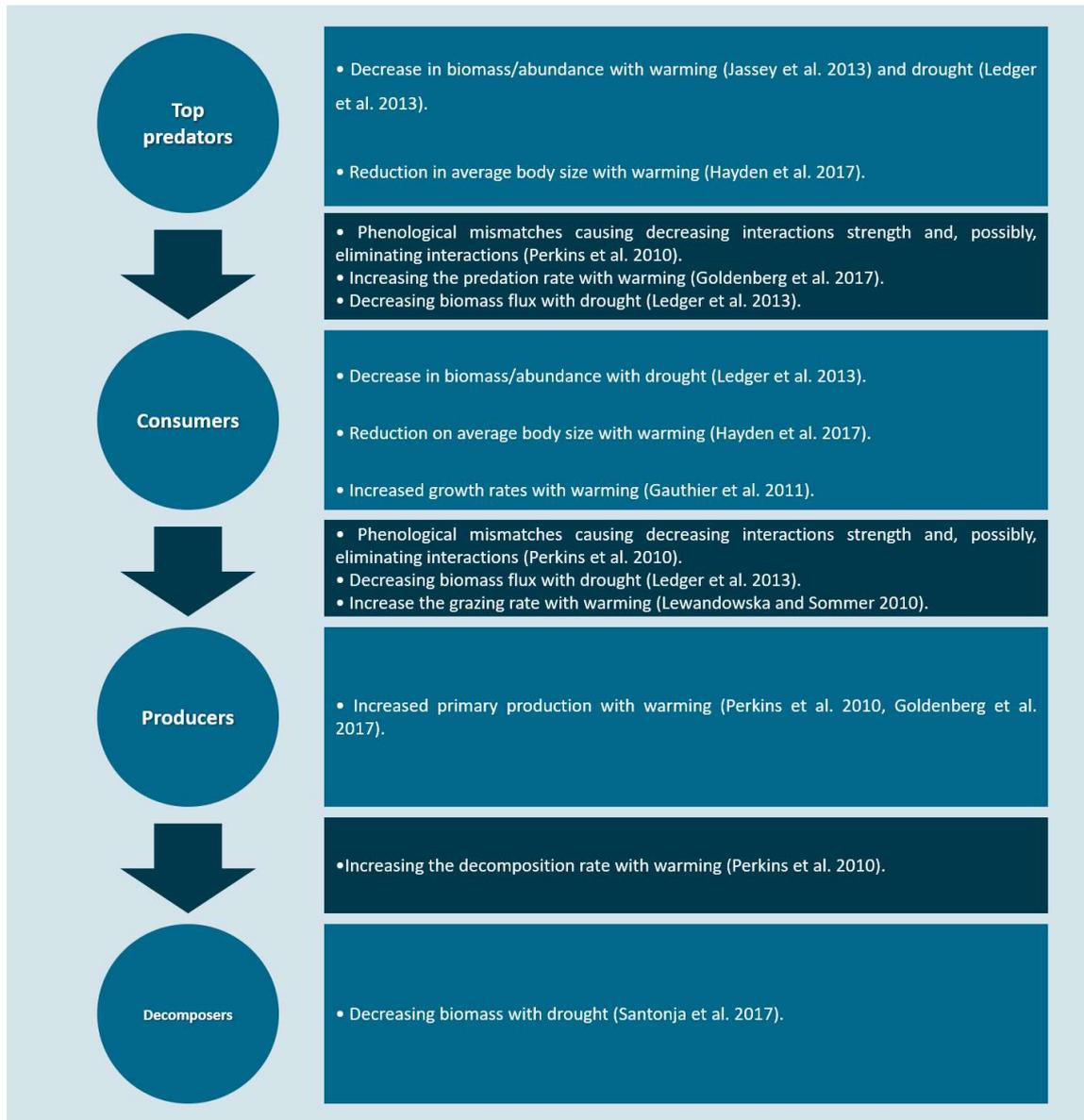
135 Figure 2 – The number of articles assessing the significant relations between environmental and human-related
136 drivers (columns) and response variables (rows).

137 The effects of environmental warming on food webs, and consequently ecosystem
138 functioning, is a concern that is well demonstrated by the frequency with which this
139 variable is addressed in research (35.2% of the articles consider either temperature or
140 temperature variability to be significant predictors).

141 The presence of other species, such as predators (e.g. Maran & Pelini 2016) or species
142 at the base of the food web (Charvet et al., 2014), another variable gathering attention
143 from researchers in the last decade (26.1% of the articles consider biotic predictors to
144 be significant predictors). With the ongoing environmental change, and the consequent
145 range shifts and local extinctions, communities are expected to undergo compositional
146 changes, triggering the emergence of novel communities (e.g. Lurgi et al. 2012). These
147 changes, a consequence of communities suffering local extinctions or species inputs,
148 will impact food webs (e.g. Seifert et al. 2015) and have been the focus of much
149 attention in the last decade, as demonstrated by the high frequency of articles resorting
150 to biotic predictors.

151 The most widely reported patterns in the literature are the reduction in body size with
152 increasing temperature in most trophic levels, phenological mismatches affecting
153 interactions, increasing decomposition rate and primary productivity with increasing
154 temperature (Figure 3). However, even these patterns are not widespread and are
155 heavily dependent upon local conditions, species involved, and scale. On the other
156 hand, some of these food-web traits are interrelated, such as the body size of
157 interacting species and the interaction strength, with interaction strength being
158 positively related to the predator-prey size ratio (e.g. Legagneux et al., 2014).

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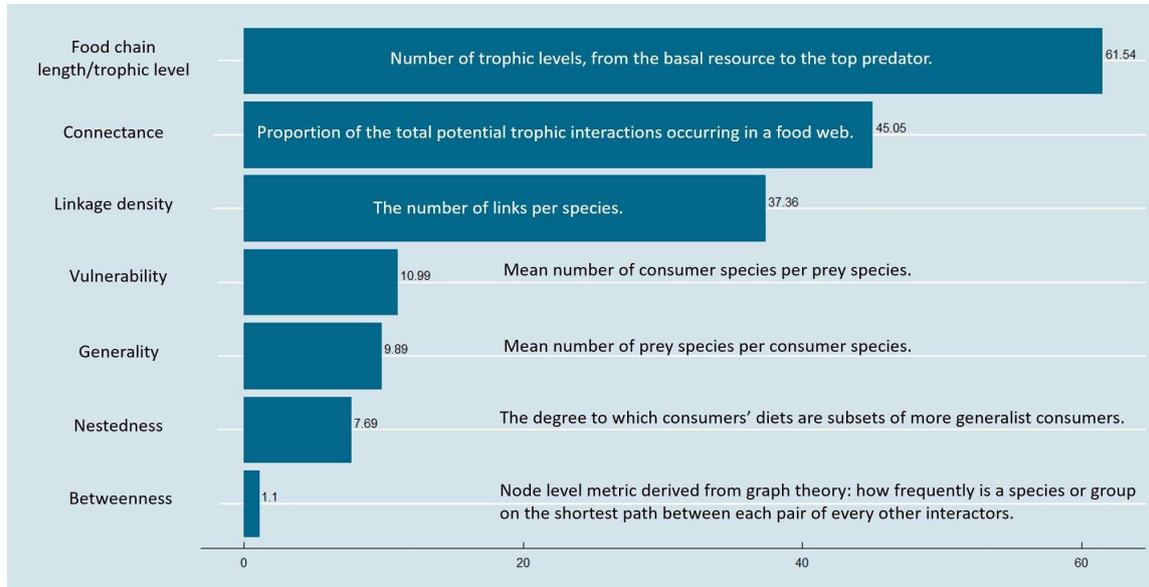
161 Figure 3 – Commonly reported environmental effects on food webs per trophic level and examples of supporting
 162 studies. The environmental effects reported in rows with dark blue arrows relate to interactions between adjacent
 163 trophic levels.

164

165 2.2. *Effects on structural metrics*

166 Collective properties of food webs can be summarized by network-level metrics. We
 167 examined the frequency with which different food web metrics were used in the
 168 reviewed articles. Only a few studies resort to network metrics to describe food webs
 169 (19.7%, 91 studies). This may be because not all studies describe food webs as
 170 networks. Some resort to describing the effects of environmental drivers on

171 interactions and species or trophic/functional groups, without requiring the use of a
 172 network structure. Food chain length, link density and connectance were the most used
 173 structural metrics (Figure 4), and thus we discuss these in greater detail in the next
 174 sections.



175

176 Figure 4 - Overview of the food-web metrics used in the reviewed literature. Percentage of each metric used in the
 177 91 studies resorting to any of the food web metrics (19.65% of the total reviewed).

178

179 **2.2.1. Food chain length**

180 It has been postulated that food chain length decreases with increasing disturbance and
 181 increases with increasing energy availability and ecosystem size (Pimm, 1991; Young et al., 2013). The literature shows that flow regulation (Ruhí et al., 2016), increased
 182 productivity (Young et al., 2013), and increased habitat size (Baiser et al., 2012) increase
 183 food chain length. These proposed relationships are intrinsically related to energy
 184 availability since, all other things being equal, a greater area corresponds to a greater
 185 amount of energy available. Likewise, an increased disturbance corresponds to a
 186 decrease in the constancy with which energy is available to organisms. Yet studies do
 187 not treat disturbance, energy availability, and ecosystem size as equivalent, rather
 188 searching for the best mutually exclusive correlates. Disturbance has been proposed as
 189 a candidate driver for food chain length in systems prone to disturbance (e.g.
 190 freshwater systems affected by changes in hydrology, Ruhí et al. (2016)). In systems
 191

192 characterized by greater levels of environmental stability, habitat size is another
193 candidate driver of food chain length (Baiser et al., 2012). In systems with
194 environmental drivers directly limiting primary producers, such as nutrient availability
195 (Doi, 2012), resource availability appears to be the main driver.

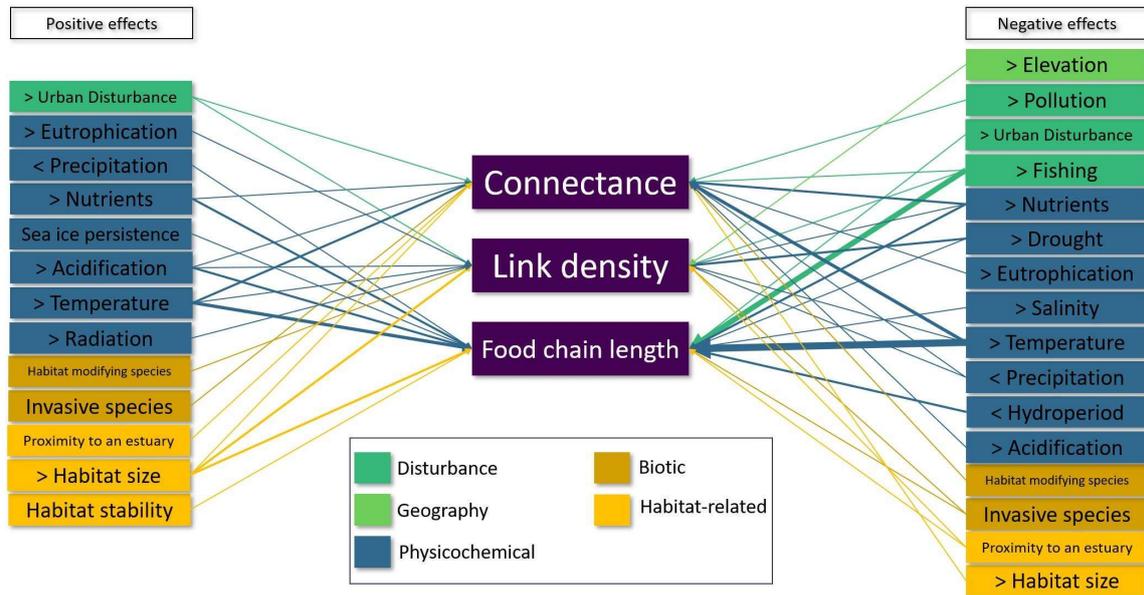
196 2.2.2. *Connectance*

197 Connectance characterizes food web complexity (Poisot & Gravel, 2014) and has been
198 related to community stability, robustness (De Angelis, 1975; Dunne et al., 2002), and
199 species richness (Warren, 1990). It is still not settled whether increased connectance
200 increases or decreases network resilience (De Angelis, 1975; Dunne et al., 2002;
201 Gardner & Ashby, 1970), or whether species richness covaries with connectance
202 (Warren, 1990; Winemiller, 1989). However, a recent study using 116 empirical food
203 webs concluded that stability might not be related to traditional descriptors of
204 connectivity, such as connectance (Jacquet et al. 2016). In the present review, we found
205 a few examples of studies relating complexity and stability (e.g. Garay-Narvaez et al.
206 2013, Galiana et al. 2014, Cesarz et al. 2015). For example, in a study performing *in*
207 *silico* experiments on empirical vertebrate food webs (Galiana et al., 2014), researchers
208 concluded that those with lower connectance were shown to be less resistant to
209 invasions, probably because vacant niches are more likely.

210 Connectance has also been related to ecosystem disturbance and variability. For
211 example, urbanization (Docile et al., 2016) and proximity to a river estuary in coastal
212 food webs (Careddu et al., 2015) have been shown to relate to increased connectance.
213 In both cases, the lower number of species found in the more urbanized sites and near
214 the river estuary might have caused connectance to increase, consistent with the
215 hypothesized inverse relationship between connectance and species richness (May,
216 1972). Finally, connectance is relatively robust to differences in sampling effort
217 (Martinez et al., 2012), taxonomic resolution (Martinez, 1993), and scale (Martinez &
218 Havens, 1993), which makes it a reliable metric for comparing multiple studies which
219 consider a diversity of node resolutions and spatial and temporal scales.

220 In our review, the effect of each environmental gradient on the three most common
221 metrics was frequently inconsistent. We identified, for example, both positive and

222 negative effects of increasing temperature on connectance, link density and food chain
 223 length (Figure 5).



224

225 Figure 5 – Main significant environmental effects on connectance, link density, and food chain length found in the
 226 literature review. Arrow thickness expresses the number of articles on which it is based. The > and < sign express the
 227 directionality in the environmental gradient. Connections are scaled by the number of articles in the database.

228 As such, one of the most pervasive findings is that studies particularities hinder the
 229 emergence of general patterns.

230

231 3. The need for a uniform approach

232 While food webs can be clearly defined as "... the feeding relationships among species
 233 or groupings of species" (Moore & De Ruiter, 2012), there is substantial room for
 234 interpretation regarding the specifics of the relationships. How should interacting
 235 species or groups of species be defined? Should nodes have the same taxonomic or
 236 functional resolution? How should interactions be measured? How are spatial
 237 boundaries of food webs determined? How should the relevant temporal resolution
 238 and extent of the interactions be defined? Different answers to these questions will
 239 affect the study design, results, and interpretations of underlying patterns and
 240 processes in food webs. This variability, as we have shown, is hindering the

241 comparability and potential generalizations about environmental effects on food webs
242 structure and function (e.g. Martinez 1991, Dunne 2005).

243 *3.1. Defining the food-web: its elements and interactions*

244 Studies use food webs with varying resolutions for nodes and different methods to
245 determine interactions (Supporting Information Figures A5-6). Node resolution (species,
246 taxonomic, trophic, and functional groups) influences structural metrics like
247 connectance, linkage density or complexity, and predator:prey ratios (Thompson &
248 Townsend, 2000). It can vary between and within food web studies, with broader
249 taxonomic classifications being common at basal trophic levels and finer resolutions
250 being common at higher levels. Most of the articles reviewed here use the broadest
251 possible class, "other taxonomic groups" (47.7% of the total number of articles). The
252 recurrence of coarse taxonomic resolutions in food web studies likely is a consequence
253 of the basal elements of the food webs being more taxonomically aggregated
254 (Supporting Information Figures A6). It follows that food web metrics vary in their
255 sensitivity to the aggregation of species into functional or trophic groups. Connectance
256 and predator:prey ratio are almost invariant to aggregation (Martinez, 1993). For
257 instance, one study (Sugihara et al. 1997) concluded that only the proportion of basal
258 species and linkage density are affected by aggregation. This study also suggested that
259 metrics are more susceptible to taxonomic aggregation than trophic aggregation
260 because trophic aggregation lumps together functionally similar species, leading to a
261 smaller impact on the overall structure of the network.

262 Trophic interactions can be characterized by energy fluxes, frequency of feeding events,
263 impact on growth rates or impact on population sizes (Berlow et al., 2004). Detection of
264 trophic interactions frequently uses stable isotopes (14.3%, second only to using
265 previous references, 46.0%). Other methods of studying interactions include statistical
266 associations between the occurrences/abundances/biomass of the interacting species
267 (5.4%), feeding experiments (4.3%), and gut content analysis (4.1%) (Supporting
268 Information Figure A5).

269 Method choice can also impact food web structure: stable isotopes and fatty acids
270 identify feeding interactions over a wider period of time (time-averaged results)

271 providing information even if the stomachs are empty (Kolts et al., 2013); stomach
272 contents provide greater taxonomic resolution but are invasive (Kolts et al., 2013).

273 The variety of approaches in use inevitably limits comparability across studies and
274 compromises generalizations. Further research is needed on how to standardize
275 interaction measurements from networks estimated using different sampling
276 methodologies.

277 *3.2. Food-webs in space and time*

278 Food webs typically characterize snapshots of trophic interactions at a given place and
279 time. However, since studies vary in the spatial and temporal resolutions and extents
280 used to characterise them (e.g., spatial scales ranging from “local” to “global”, or
281 temporal scales from “< 1 year” to “> 10 years”), detection of general patterns is most
282 likely impaired (Supporting Information Figure A7-A9).

283 The spatial boundaries delimiting the sources of data used to construct food webs are
284 often arbitrary (Baiser et al., 2012) and vary with observer perceptions or conveniences
285 (Moore & De Ruiter, 2012). Additionally, according to Cohen (1978), food webs can be
286 divided into source, sink, and community food webs. These differences in determining
287 which species are considered have implications for determining the spatial boundaries
288 of the food web. Some examples are present in the literature reviewed, such as the
289 detritus-based source food web in Lake Obersee, Germany (Majdi et al, 2016), the sink
290 food web focused on the minke whale in the Barents Sea (Lindstrom et al., 2009) or the
291 freshwater community at the Bere Stream, England (Woodward et al., 2008). Spatial
292 boundaries can be determined based on organismal home ranges or, more generally,
293 the physical limits to movement (Moore & De Ruiter, 2012) (e.g., chalk stream food
294 webs in southern England, Woodward et al. 2008). Food webs sampled across
295 increasingly large extents are more likely to lose information regarding the actual
296 trophic interactions, instead of representing potential interactions (e.g. Braga et al.
297 2019). Comparisons across food web topologies measured at different scales thus need
298 to be done with caution.

299 Food webs are also not spatially isolated, being connected to external elements
300 through dispersal or allochthonous subsidies (Massol et al., 2011; Meunier et al., 2017).
301 They can be interconnected through dispersal if, for instance, a predator with large
302 home ranges connects otherwise disconnected local food-webs (e.g. McCann et al.
303 2005). On the other hand, allochthonous subsidies (Meunier et al., 2017), whether at
304 the bottom or elsewhere in the food web, can constitute major energy sources,
305 sustaining complex organizations. Both dispersal and allochthonous subsidies make the
306 definition of food web boundaries difficult to establish.

307 Spatial scale has two main components, namely extent (the area of the study site) and
308 resolution (the resolution of the food webs). As an example, one article evaluating food
309 web richness and composition variability across 39 sites in the United States (Buckley et al.,
310 2010) was classified as having a “continental” spatial extent, but a “local”
311 resolution. The studies reviewed here were mostly local, whether in extent or
312 resolution (Supporting Information Figure A7-8). Most studies dwell on local food webs,
313 on small-sized systems (e.g., microcosms, mesocosms, experimental field sites, lakes)
314 (66.3%), several studies have a regional scale (25.1%), and only a few refer to the global
315 (2.2%) or continental/oceanic scales (1.9%). Spatial resolution is, as expected, even
316 more dominated by local (77.1%) and regional scales (18.8%).

317 Most studies on food webs do not provide replicates, whether spatial or temporal, or
318 resort to temporal and spatial averaging. Environmental variation, changes in
319 abundance, and observation errors can cause variation in pairwise interactions and in
320 characterizations of food web metrics (Cirtwill et al., 2019). It follows that uncertainty
321 in the detection of pairwise interactions scales up to the network level potentially
322 biasing some structural properties more than others (Poisot et al., 2015). Communities
323 are notoriously variable, and it is standard practice to evaluate and consider uncertainty
324 in comparative studies. However, probably owing to difficulties in documenting
325 pairwise interactions and characterising the full spectrum of interactions in a network,
326 attempts to characterise food webs often result in a single characterisation, a snapshot,
327 with no attempts to assess their representativeness. A few studies have some way of
328 including spatial variability in food web structure (27.4% of the total), generally through
329 multiple sampling sites within the study area, although spatially averaging the resulting

330 food web. Some go a step further and effectively evaluate the spatial heterogeneity
331 along environmental gradients. For instance, Doi and colleagues (2013) demonstrated
332 small-scale spatial heterogeneity among planktonic food webs along an environmental
333 gradient of water chemistry and primary productivity. In a study where the spatial scale
334 and extent were varied systematically, the latter was found to have more impact on
335 network metrics considering the relevance of fully encompassing the environmental
336 gradients considered (Vinagre et al, 2017). Around a third of the studies account for
337 temporal variability in food web structure, at least by averaging the sampling at
338 multiple time points (temporal averaging) or considering multiple sampling time points
339 (e.g. seasons) (32.18% of the total). Temporal averaging is likely to mask seasonal or
340 inter-annual dynamics, affecting the structural metrics of local food webs (Jordán &
341 Osváth, 2009; López et al., 2018). The majority of studies on temporal dynamics of
342 networks concern mutualistic interactions (e.g. CaraDonna et al. 2017), with just a few
343 dealing with food webs. A study, conducted at the Åland Islands in the Northern Baltic
344 Sea, evaluated the temporal variability in a benthic food-web (Nordström et al., 2009)
345 and concluded that there is intra-seasonal variation in food web components both
346 within and between trophic levels. Additionally, in this study, interannual variability was
347 limited, demonstrating that the food web was temporally stable. However, seasonal
348 fluctuation of stable isotope values decreased in the higher trophic level which shows
349 that distinct trophic levels respond differently to time, which is certainly a consequence
350 of the fact that lower trophic levels have generally shorter life spans and faster
351 generational times. Consequently, a difference in the food webs' basal resources might
352 be diluted in the upper trophic levels (Kolts et al., 2013; O'Reilly et al., 2002).

353 Most of the reviewed studies include periods of less than one year (46.4%) or more
354 than ten years (24.0%) and just a few address periods of one to five (18.8%) and five to
355 ten years (4.1%) (see also Supporting Information Figure A9). The relatively short time
356 span of temporal food-web studies reduces dramatically the likelihood of detecting
357 responses to sub-lethal disturbances, which may take decades to express themselves
358 (e.g. Peterson et al. 2003).

359 *3.3. Diversity of approaches and objectives*

360 There is no standardized approach to evaluate the impacts of environmental variables
361 on food webs and, in most studies, this is not even the primary objective. Several
362 studies resort to the comparison of food web structure across gradients, like
363 temperature (e.g. Franzè & Lavrentyev 2017) or between sites with different
364 environmental characteristics (e.g. Matias et al. 2017) (observational studies: 38.9%).
365 Others use experimental manipulation of environmental characteristics in mesocosms
366 (e.g. Özen et al. 2013), microcosms (e.g. Burgmer & Hillebrand 2011) or simulate
367 natural food webs in virtual environments (e.g. Zhang et al. 2017) (controlled
368 experiments: 32.6%; natural experiments: 11.5%; simulation 5.8%).

369 Studies also vary in their objectives. Marine studies (43.6%, see Fig.1) are dominated by
370 fishing-related management problems, such as sustainability (Lindegren et al., 2009),
371 top predator overfishing (Llope et al., 2011), or fish stock evaluation (Kempf et al.,
372 2006). The objectives of freshwater studies (28.9%) are more diverse, with some
373 addressing issues like the impact of water mixing (Blottière et al., 2017), the effect of
374 temperature (Zander et al., 2017), light availability (Collins et al., 2016), salinity
375 (Cañedo-Argüelles et al., 2016), drought (Lu et al., 2016), flow regulation (Ruhí et al.,
376 2016), or the variation in decomposition rates (Ferreira et al., 2015). Terrestrial studies
377 (12.7%) tend to focus on the effects of environmental factors, such as precipitation
378 (Deguines et al., 2017), temperature (Sentis et al., 2013), CO₂ (Dyer et al., 2013), or
379 climate change as a whole (Mortensen et al., 2016). Finally, articles on soil food webs
380 (11.0%) address mainly the relations with plants (Cesarz et al., 2017), allochthonous
381 nutrient input (Hu et al., 2017), elevated CO₂ (Mueller et al., 2016), hydrological
382 changes (Sun et al., 2016) and warming (Schwarz et al., 2017).

383

384 **4. Progresses made in the last decade**

385 Previous authors have made recommendations to advance food web ecology focusing
386 mainly on data quality and the move towards more predictability (e.g. Cohen et al.
387 1993, Dunne 2005, Ings et al. 2009, Thompson et al. 2012). It is important to reflect on
388 how far we have gone and if we are moving in the right direction.

389 A positive sign is an emergence of manipulative studies (e.g., mesocosms) or natural
390 experiments, as a more generalized approach. However, observational studies still
391 dominate, representing most of the reviewed literature. Despite progress with
392 techniques allowing better descriptions of food-web structure, such as stable isotopes
393 or fatty acids to determine interactions or environmental DNA to identify the species
394 present (Charvet et al., 2014), their use is still not widespread. Researchers are heavily
395 reliant on published information as the main source of information to determine
396 interactions and node resolution is still defined in broader taxonomic resolutions than
397 the species.

398 Another unresolved issue is the lack of data standardization, of which our review only
399 shows modest signs of progress. One notable exception is the many marine models
400 developed with Ecopath with Ecosim (Christensen & Walters, 2004), which require the
401 data and metadata to be standardized and stored in an online database with a strictly
402 defined structure.

403 Some of the reviewed articles seek a more mechanistic, and thus more predictive
404 approach to understanding the way food webs respond to environmental gradients, as
405 recommended by earlier reviews (e.g. Ings et al. 2009). Understanding the underlying
406 processes explaining food web structural response to gradient is achieved by resorting
407 to experimental manipulation. Many of the reviewed studies are controlled
408 manipulative or natural experiments as mentioned above.

409 Previous works also called for a stronger theoretical framework (e.g. Ings et al. 2009),
410 for example by integrating food web ecology with foraging theory or the metabolic
411 theory of ecology. Some of the reviewed articles seek such an intersection between the
412 metabolic theory of evolution and food web ecology (Eklöf et al., 2012; Sentis et al.,
413 2014), but many articles are still focused on local or regional research questions of
414 applied value for the management of natural resources (e.g., fisheries) without seeking
415 a connection to a broader unifying ecological theory.

416 Past work (Cohen et al., 1993) has also highlighted the need for collaboration between
417 researchers to tackle the challenge of identifying multiple taxa, bringing in taxonomists
418 to work with food web ecologists. The need for geographically distributed coordinated
419 experimental studies is an additional incentive to collaboration, considering it involves
420 researchers working on multiple regions.

421 **5. Perspectives**

422 Despite hundreds of empirical and experimental studies in the past decades, food-web
423 ecology still lacks general predictions about environmental effects on trophic
424 interactions and the associated effects on ecosystem function. Indeed, no ubiquitous
425 relationships between food-web structural properties and environmental gradients
426 emerge from our review despite recent studies, building on the concept of meta-webs
427 (Morales-Castilla et al., 2015), uncovering clear relationships with environmental
428 parameters (Mendoza & Araújo, 2019) or relating the number of trophic levels with
429 productivity, and temperature (both, average and seasonality) (Oksanen et al., 2020).
430 Despite progress in sampling techniques (e.g., stable isotopes or environmental DNA),
431 there is substantial scope for improving and unifying methodologies and the conceptual
432 underpinnings of much of the research. Developments in three areas of inquiry could
433 help the field move forward.

434 *5.1. Refinement of theory and testable predictions*

435 Science is usually on the right path when small sets of testable predictions on the most
436 pressing questions are agreed upon and pursued collectively by researchers in the field.
437 Food-web ecology still lacks such a tight framework. There are pieces in the literature
438 exploring how aspects of food-web structure vary over gradients (Pellissier et al., 2017;
439 Tylianakis & Morris, 2017). For instance, there is a long tradition to study how trophic
440 regulation varies over productivity (e.g., the green world hypothesis, Hairston et al.
441 (1960)). Recently, a quantitative framework has been proposed to partition the drivers
442 of network change at the biogeographical scales (Gravel et al., 2018; Poisot et al.,
443 2015). The integration of metabolic theory to consumer-resources theory (Brown et al.,
444 2004) also allows investigating how pairwise interactions and trophic regulation scale
445 with temperature (Bideault et al., 2019; Gounand et al., 2016). Another example is the
446 integration of optimal foraging theory to predict food web structure (Petchey et al.,
447 2008). However, akin to empirical observations, theory is fragmented by the diversity of
448 gradients and models used to study food-web properties.

449 A consistent theory of food-web dynamics across environmental gradients will require a
450 scalable approach, with comparable models to the study of food-web modules (to

451 understand mechanisms) up to the study of entire networks (to document emerging
452 properties). It should focus on a few gradients, and we note that temperature, solar
453 radiation intensity, and primary productivity are likely the best candidates because of
454 their relevance as surrogates of energy availability. These gradients are intimately
455 related, although their effects might differ since temperature directly affects all species
456 in the food-web via physiological effects (Brown et al., 2004), while radiation and
457 productivity affect plants directly and indirectly higher trophic levels (Loreau, 2010).
458 Similarly, theory should focus on the set of food-web functional properties that are
459 measurable in the field and relevant to understand ecosystem functioning. Based on
460 tradition and relevance for global change studies, a useful path is to focus on biomass
461 distribution, consumption:production ratio, trophic regulation and network topology.
462 Lastly, we emphasize that the development of theory should be performed in
463 conjunction with experiments and observations (see below) so that quantitative
464 predictions can be formulated and adequately tested.

465 *5.2. Minimum sets of agreed standards for data collection and analysis*

466 Studies of food webs can dwell on different questions, and although refinement of
467 theories and hypotheses is expected to reduce variation in the methodologies for data
468 acquisition and analysis (see above), there will always be diversity in the approaches
469 used.

470 A consistent definition of “interaction” is required. Variation in criteria for link
471 determination could limit comparability with researchers casting doubt on the
472 usefulness or even validity of binary depictions of food-web interactions (Banašek-
473 Richter et al., 2004). There is a range of direct and indirect approaches for identifying
474 links and/or measuring strengths of interactions in food-webs and there is no guarantee
475 that the conclusions with different methods are comparable (Berlow et al., 2004).
476 Agreement on a core set of procedures for establishing such links would help reduce
477 variation in food-web patterns associated with variation in the methods used.
478 Furthermore, food webs should be defined by natural spatial boundaries constraining
479 the movement of a majority of organisms. Finally, the taxonomic resolution of the
480 nodes should be increased (ideally at the species level) and minimally it should be

481 consistent within a single food-web. Hopefully, novel molecular methods to process gut
482 contents will solve part of the problem ensuring species-level resolution of the nodes
483 (Roslin et al., 2016).

484 Nevertheless, minimum agreed standards are needed for enabling comparability across
485 studies (Poisot et al., 2016), an effort already achieved for using data and models of
486 species distributions (Araújo et al., 2019). In particular, attention should be given to
487 robust data specifications to aggregate datasets and perform comparative studies. The
488 “mangal” data specification (Poisot et al., 2016), for instance, is a common language to
489 store and share network data. It is based on a hierarchical collection of ecological
490 objects, from individuals to networks. The adoption of such a standard not only
491 requires taking into consideration data acquisition and structure, it also implies that all
492 relevant metadata is collected, making comparative studies over gradients more
493 accessible.

494 *5.3. Development of geographically distributed experimental studies*

495 Understanding how food webs respond to perturbations relies heavily on the ability to
496 generalize conclusions often derived from observations at single locations. The need for
497 geographical replication of local experiments across environmental gradients is critical
498 for understanding the relationship between biological observations and environmental
499 or evolutionary predictors (Freestone & Osman, 2011; Pelini et al., 2014). The
500 implementation of multiple-site experiments can lend support to the interpretation
501 that local observations are not the product of local contingencies, rather providing
502 meaningful inferences across scales (Borer et al., 2014). Examples of geographically
503 distributed experiments include BIODDEPTH, probably the first large-scale coordinated
504 experiments so far, designed to test the relationships between biodiversity and
505 ecosystem functions (Hector et al., 1999). The development of globally replicated
506 experiments assessing the responses of food webs to perturbations across relevant
507 environmental gradients has the potential to establish a benchmark for the future
508 development of predictive food-web models.

509

510

511 **6. Conclusions**

512 The analytical tools required and the setting up of manipulative experiments extending
513 across large geographical areas and periods of time is a costly endeavour requiring
514 collaboration between researchers across regions. Advancing food web ecology is
515 poised with several scientific challenges, logistical difficulties, and lack of funding. We
516 encourage researchers to develop theoretical and methodological approaches, as well
517 as geographically replicable sampling schemes, to help solve these long-standing issues
518 concerning the environmental drivers coercing food-web structure.

519

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