

Predicting metawebs: graph embeddings can help alleviate spatial data deficiencies

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, i.e. networks of potential interactions within a species pool, are a powerful abstraction to understand how large-scales species interaction networks are structured. metawebs are typically expressed at large spatial and taxonomic scales, assembling them is a tedious and costly process; predictive methods can help circumvent the limitations in data deficiencies, by providing 'draft' metawebs. way to improve the predictive ability is to maximize the information used for prediction, by using graph embeddings rather than the list of species interactions. Graph embedding is an emerging field in machine learning that holds great potential for ecological problems. this perspective, we outline how the challenges associated with inferring metawebs line-up with the advantages of graph embeddings; furthermore, because metawebs are inherently spatial objects, we discuss how the choice of the species pool has consequences on the reconstructed network, but also embeds hypotheses about which human-made boundaries are ecologically meaningful.

1 Having a general solution for inferring *potential* interactions (despite the unavailability of interaction data)
2 could be the catalyst for significant breakthroughs in our ability to start thinking about species interaction
3 networks over large spatial scales. In a recent overview of the field of ecological network prediction,
4 Strydom et al. (2021) identified two challenges of interest to the prediction of interactions at large scales.
5 First, there is a relative scarcity of relevant data in most places globally – paradoxically, this restricts our
6 ability to infer interactions to locations where inference is perhaps the least required; second, accurate
7 predictions often demand accurate predictors, and the lack of methods that can leverage small amount of
8 data is a serious impediment to our predictive ability globally.

9 Following the definition of Dunne (2006), a metaweb is a network analogue to the regional species pool;
10 specifically, it is an inventory of all *potential* interactions between species from a spatially delimited area
11 (and so captures the γ diversity of interactions). The metaweb is, therefore, *not* a prediction of the food
12 web at a specific locale within the spatial area it covers, and will have a different structure (notably by
13 having a larger connectance; see e.g. Wood et al., 2015). These local food webs (which captures the α
14 diversity of interactions) are a subset of the metaweb’s species and interactions, and have been called
15 “metaweb realizations” (Poisot et al., 2015). Differences between local food web and their metaweb are
16 due to chance, species abundance and co-occurrence, local environmental conditions, and local
17 distribution of functional traits, among others.

18 Because the metaweb represents the joint effect of functional, phylogenetic, and macroecological
19 processes (Morales-Castilla et al., 2015), it holds valuable ecological information. Specifically, it is the
20 “upper bounds” on what the composition of the local networks can be (see e.g. McLeod et al., 2021). These
21 local networks, in turn, can be reconstructed given appropriate knowledge of local species composition,
22 providing information on structure of food webs at finer spatial scales. This has been done for example for
23 tree-galler-parasitoid systems (Gravel et al., 2018), fish trophic interactions (Albouy et al., 2019), tetrapod
24 trophic interactions (O’Connor et al., 2020), and crop-pest networks (Grünig et al., 2020). Whereas the
25 original metaweb definition, and indeed most past uses of metawebs, was based on the presence/absence
26 of interactions, we focus on *probabilistic* metawebs where interactions are represented as the chance of
27 success of a Bernoulli trial (see e.g. Poisot et al., 2016); therefore, not only does our method recommend
28 interactions that may exist, it gives each interaction a score, allowing us to properly weigh them.

29 Yet, owing to the inherent plasticity of interactions, there have been documented instances of food webs
30 undergoing rapid collapse/recovery cycles over short periods of time (Pedersen et al., 2017). The

31 embedding of a network, in a sense, embeds its macro-evolutionary history, especially as RDPG captures
 32 ecological signal (Dalla Riva & Stouffer, 2016); at this point, it is important to recall that a metaweb is
 33 intended as a catalogue of all potential interactions, which should then be filtered (Morales-Castilla et al.,
 34 2015). In practice (and in this instance) the reconstructed metaweb will predict interactions that are
 35 plausible based on the species' evolutionary history, however some interactions would/would not be
 36 realized due to human impact.

37 Dallas et al. (2017) suggested that most links in ecological networks may be cryptic, *i.e.* uncommon or
 38 otherwise hard to observe. This argument essentially echoes Jordano (2016): the sampling of ecological
 39 interactions is difficult because it requires first the joint observation of two species, and then the
 40 observation of their interaction. In addition, it is generally expected that weak or rare links would be more
 41 common in networks (Csermely, 2004), compared to strong, persistent links; this is notably the case in
 42 food chains, wherein many weaker links are key to the stability of a system (Neutel et al., 2002). In the
 43 light of these observations, the results in fig. ?? are not particularly surprising: we expect to see a surge in
 44 these low-probability interactions under a model that has a good predictive accuracy. Because the
 45 predictions we generate are by design probabilistic, then one can weigh these rare links appropriately. In a
 46 sense, that most ecological interactions are elusive can call for a slightly different approach to sampling:
 47 once the common interactions are documented, the effort required in documenting each rare interaction
 48 may increase exponentially. Recent proposals suggest that machine learning algorithms, in these
 49 situations, can act as data generators (Hoffmann et al., 2019): in this perspective, high quality
 50 observational data can be supplemented with synthetic data coming from predictive models, which
 51 increases the volume of information available for inference. Indeed, Strydom et al. (2021) suggested that
 52 knowing the metaweb may render the prediction of local networks easier, because it fixes an “upper
 53 bound” on which interactions can exist; indeed, with a probabilistic metaweb, we can consider that the
 54 metaweb represents an aggregation of informative priors on the interactions.

55 As Herbert (1965) rightfully pointed out, “[y]ou can’t draw neat lines around planet-wide problems”; in
 56 this regard, our approach (and indeed, any inference of a metaweb at large scales) must contend with
 57 several interesting and interwoven families of problems. The first is the limit of the metaweb to embed
 58 and transfer. If the initial metaweb is too narrow in scope, notably from a taxonomic point of view, the
 59 chances of finding another area with enough related species to make a reliable inference decreases; this
 60 would likely be indicated by large confidence intervals during ancestral character estimation, but the lack

61 of well documented metawebs is currently preventing the development of more concrete guidelines. The
62 question of phylogenetic relatedness and dispersal is notably true if the metaweb is assembled in an area
63 with mostly endemic species, and as with every predictive algorithm, there is room for the application of
64 our best ecological judgement. Conversely, the metaweb should be reliably filled, which assumes that the
65 S^2 interactions in a pool of S species have been examined, either through literature surveys or expert
66 elicitation. Supp. Mat. 1 provides some guidance as to the type of sampling effort that should be
67 prioritized. While RDPG was able to maintain very high predictive power when interactions were missing,
68 the addition of false positive interactions was immediately detected; this suggests that it may be
69 appropriate to err on the side of “too many” interactions when constructing the initial metaweb to be
70 transferred. The second series of problems are related to determining which area should be used to infer
71 the new metaweb in, as this determines the species pool that must be used.

72 In our application, we focused on the mammals of Canada. The upside of this approach is that
73 information at the country level is likely to be required by policy makers and stakeholders for their
74 biodiversity assessment, as each country tends to set goals at the national level (Buxton et al., 2021) for
75 which quantitative instruments are designed (Turak et al., 2017), with specific strategies often enacted at
76 smaller scales (Ray et al., 2021). And yet, we do not really have a satisfying answer to the question of
77 “where does a food web stop?”; the current most satisfying solutions involve examining the spatial
78 consistency of network area relationships (see e.g. Galiana et al., 2018, 2019, 2021; **Fortin2021NetEco?**),
79 which is of course impossible in the absence of enough information about the network itself. This
80 suggests that an *a posteriori* refinement of the results may be required, based on a downscaling of the
81 metaweb. The final family of problems relates less to the availability of data or quantitative tools, and
82 more to the praxis of spatial ecology. Operating under the context of national divisions, in large parts of
83 the world, reflects nothing more than the legacy of settler colonialism. Indeed, the use of ecological data is
84 not an apolitical act (Nost & Goldstein, 2021), as data infrastructures tend to be designed to answer
85 questions within national boundaries, and their use both draws upon and reinforces territorial statecraft;
86 as per Machen & Nost (2021), this is particularly true when the output of “algorithmic thinking” (e.g.
87 relying on machine learning to generate knowledge) can be re-used for governance (e.g. enacting
88 conservation decisions at the national scale). We therefore recognize that methods such as we propose
89 operate under the framework that contributed to the ongoing biodiversity crisis (Adam, 2014), reinforced
90 environmental injustice (Choudry, 2013; Domínguez & Luoma, 2020), and on Turtle Island especially,

91 should be replaced by Indigenous principles of land management (Eichhorn et al., 2019; No'kmaq et al.,
92 2021). As we see AI/ML being increasingly mobilized to generate knowledge that is lacking for
93 conservation decisions (e.g. Lamba et al., 2019; Mosebo Fernandes et al., 2020), our discussion of these
94 tools need to go beyond the technical, and into the governance consequences they can have.

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