

Beyond Species Richness: Empirical Assessment of a Consumption-Weighted Functional Diversity Index Across Aquatic Ecosystems

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Lower Jinsha River | Singapore reservoirs | Coweeta experimental streams

Abstract

Conventional taxonomic indices describe community composition but do not explicitly integrate the distribution of resource consumption with differences in dietary niche use. The Functional Diversity Index (FDI), previously introduced in a coauthored NetLogo simulation study (Ayyanger and Vijay, 2026), combines these dimensions through a consumption-weighted quadratic formulation. This study evaluates that previously proposed index using temporal, spatial, and experimental aquatic food-web datasets. Its purpose is to determine what FDI describes in empirical communities, how its interpretation depends on measurement choices, and whether it improves prediction of an independently measured ecosystem process.

Three datasets provide complementary tests of the framework. In the lower Jinsha River, the period with the highest fish Shannon diversity had the lowest consumption-weighted FDI (0.05516), and replacing consumption shares with biomass shares changed which period ranked highest. Across 12 Singapore reservoirs, fish-only dietary dissimilarity varied among reservoir types (permutation $p = 0.0366$), whereas the whole-food-web comparison did not (Kruskal–Wallis $p = 0.8508$). Absolute consumer–consumption estimates were unavailable, so the complete FDI could not be calculated for these reservoirs. In five experimental streams, total consumption increased during the enrichment years while FDI changed in different directions; restricting calculations to taxa present across periods preserved 8 of 10 before–after directional comparisons. Corrected exclusion of aggregate spreadsheet rows yielded 192 consumer–stream–year profiles and 15 complete stream–year estimates. In addition, an independent litter-decomposition assessment did not show improved leave-one stream-out predictive accuracy when FDI was added to a period-only model (RMSE 0.002016 versus 0.001720).

Together, these analyses establish the empirical computability of FDI where appropriate inputs exist, demonstrate its sensitivity to consumer weighting and food-web boundaries, and show how it can describe trophic patterns that differ from those summarized by taxonomic diversity or total resource flow. While the findings support the FDI as a complementary, interpretable descriptor of consumption-weighted trophic organization, they do not prove superiority in predicting ecosystem functioning.

1 Introduction

Biodiversity is more than the number of species present in an ecosystem. Two communities can contain similar numbers of species yet differ in which organisms use the available resources, how much they consume, and how strongly their diets overlap. Environmental change may alter these relationships without producing an equally clear change in species richness. For that reason, taxonomic diversity and trophic organization are related but distinct descriptions of a community.

The functional roles of organisms are not fixed. As food availability and environmental conditions change, some consumers may account for a larger share of resource use while others become less prominent. Their diets may also become more similar or more differentiated. Rao’s quadratic entropy offers a way to summarize weighted differences among organisms, and Pianka’s index describes overlap between their resource-use profiles. Yet a dietary-overlap measure by itself does not tell us whether two consumers make comparable contributions to the community’s total resource consumption. Likewise, total consumption can rise without revealing whether it is distributed more evenly or concentrated among fewer consumers.

The Functional Diversity Index (FDI) was introduced in a previous coauthored NetLogo simulation study by Ayyanger and Vijay (2026) [22]. That study combined consumption evenness with a weighted measure of dietary dissimilarity, using organism-level oxygen-consumption assignments in simulated communities. The Version 2 preprint is available at <https://doi.org/10.21203/rs.3.rs-10821215/v2>. The mathematical framework and original simulation belong to that earlier study. The present manuscript does not treat them as new results; instead, it asks how the previously proposed index behaves when applied to ecological observations and published food-web estimates.

Three datasets offer different opportunities to examine that question. The lower Jinsha River dataset combines four period-specific Ecopath food webs with fish diversity and isotope measurements. Twelve Singapore reservoirs allow a comparison of dietary differentiation among fish and across broader food webs, although their available records do not contain the absolute intake needed for complete FDI calculation. A three-year, five-stream nutrient-enrichment experiment provides estimated resource consumption for individual macroinvertebrate taxa and a separately measured litter-decomposition response. These datasets were selected for their different temporal, spatial, and experimental structures, rather than because they could be expected to favor the proposed index.

The study addresses three questions. First, does FDI provide a descriptive view of trophic organization that is not identical to taxonomic diversity or total resource consumption? Second, how much do its values and conclusions depend on the way consumer contributions and community boundaries are defined? Third, does the combined index improve prediction of a measured ecosystem process beyond a simpler model?

2 Literature Review

2.1 Taxonomic, Trait-Based, and Niche Measures

Traditional biodiversity measures begin with the composition and relative abundance of species. Species richness counts the species present, while Shannon and Simpson indices account for how individuals are distributed among those species. These measures are useful precisely because they summarize well-defined properties of an assemblage. They do not, however, directly specify which resources species consume or how their roles differ within a food web.

Functional-diversity research extends this perspective by considering differences among organisms. Rao's quadratic entropy [3] incorporates pairwise dissimilarities weighted by the contributions of community members, while Laliberté and Legendre [7] developed a distance-based approach to measuring functional diversity across multiple traits. Botta-Dukát [1] and Ricotta and Szeidl [12] further demonstrated how the choice of distances and weighting conventions shapes the interpretation of these measures. Petchey and Gaston [23] emphasized the importance of selecting traits that match the ecological question. Together, these studies provide a foundation for interpreting FDI: weighted functional differences are not new in themselves, and the value of a combined measure depends on what its inputs represent.

Dietary niche measures describe another aspect of ecological organization. Pianka's index [11] compares the distribution of resource use between consumers, while MacArthur and Levins [9] and Schoener [15] place resource overlap in the wider context of coexistence and partitioning. A high degree of overlap means that two dietary profiles are similar; it does not by itself show whether competition occurs, whether a community is healthy, or how much material each consumer processes. Because diets can shift with resource availability and disturbance [5], their interpretation also depends on when and where the community is observed.

2.2 Resource Consumption and Food-Web Organization

The movement of material and energy through food webs provides a different view of ecological function. Lindeman's trophic-dynamic framework [8] connected feeding relationships with energy transfer, and Odum [10] examined changes in ecosystem organization during development. Metabolic theory [2] and estimates of primary productivity [13] address related questions at different scales. Together, these studies motivate measurements of consumption and resource flow, but they do not make oxygen consumption, primary productivity, and food intake interchangeable. A valid application must state which quantity was measured or estimated and how it was normalized.

This distinction matters for the present study. The earlier NetLogo simulation assigned oxygen-consumption values to individual organisms [22]. In contrast, the Jinsha food webs estimate food intake from Ecopath biomass and consumption-to-biomass ratios, while the Coweeta stream records estimate resource-specific food consumed by taxa. These are different ecological quantities. The same mathematical equation can be evaluated using their relative shares, but the resulting numbers must be interpreted in the units and assumptions of each source dataset rather than treated as direct replications of the simulation.

2.3 Functional Diversity, Community Boundaries, and Empirical Assessment

Functional diversity is not a single ecological property. Villéger, Mason, and Moullot [24] distinguished functional richness, evenness, and divergence, and Mouchet et al. [25] investigated overlap and redundancy among indices. Their findings raise a practical question for FDI: does multiplying two related components offer an interpretable summary, or does it merely reweight information already contained in one of them? This study addresses that question by reporting evenness, weighted dietary dissimilarity, and the product separately, rather than treating FDI as independent of its ingredients.

Food-web boundaries also affect the answer. The reservoir data of Wilkinson et al. [18,19] allow dietary overlap to be calculated for fish alone and for larger consumer assemblages. A difference between those two scopes is ecologically meaningful as a question of measurement, but it is not necessarily evidence that the environment itself has changed. The Jinsha archive of Ren et al. [17] includes modeled consumption and diet information for aggregated consumer groups, making a full FDI calculation possible with clearly stated assumptions. The experimental-stream data of Demi et al. [20,21] provide another complete application at a finer taxonomic resolution. A separate litter-decomposition measurement from the experimental context [26] makes it possible to examine whether FDI helps explain a process outside the index calculation itself.

Resilience theory [6], research on ecosystem regime shifts [14], and statistical indicators of critical slowing down [4] provide broader reasons to investigate changes in ecological organization. They do not, however, establish that a high dietary-diversity score implies greater resilience or that an index can identify a tipping point. Such claims require a specified ecological outcome and evidence connecting the metric to that outcome. The present paper therefore treats FDI as a proposed description of trophic organization and tests predictive relevance only where an appropriate outcome is available.

2.4 Research Gap and Contribution of the Present Study

Existing taxonomic, functional-trait, and dietary measures each describe important features of ecological communities. The previously proposed FDI [22] brings consumption shares and dietary differentiation into a single calculation. What remained uncertain after its simulation-based introduction was how the index would behave with field-derived food-web data, whether the choices needed to construct those data would change its conclusions, and whether the combined score would be useful beyond its separate components.

The contribution of the present work is therefore empirical and interpretive, rather than the invention of a second index. It documents where the complete equation can be calculated, identifies when only its dietary component is supported by the available data, compares FDI with taxonomic and resource-flow summaries, and examines its sensitivity to weighting and taxon coverage. It also reports a direct test against independently measured litter decomposition, including the absence of a predictive improvement. This combination of positive descriptive findings and explicit limits is intended to clarify when FDI can be informative and what additional evidence would be needed to establish broader ecological utility.

3 Functional Diversity Index: Generalized Mathematical Framework

The previously proposed Functional Diversity Index [22] integrates multiple ecological dimensions by combining the evenness of consumption contributions with dietary niche differentiation. Instead of focusing on raw species abundance, the first term measures the relative distribution of a specified consumption quantity across organisms or consumer groups. The second term uses a quadratic-entropy structure and Pianka-based dissimilarity to describe how different the consumers' diets are, weighted by their contributions to total consumption. The proposed index is their product.

For ecological unit u , let $c_{iu} \geq 0$ be the specified consumption quantity for consumer i , with $\sum_i c_{iu} > 0$, and let r_{iku} be the share of dietary resource k in that consumer's diet. For consumers with positive consumption and a defined diet, the weights and pairwise dietary overlap are

$$p_{iu} = \frac{c_{iu}}{\sum_{j=1}^{n_u} c_{ju}}, \quad O_{iju} = \frac{\sum_{k=1}^{K_u} r_{iku} r_{jku}}{\sqrt{\sum_{k=1}^{K_u} r_{iku}^2} \sqrt{\sum_{k=1}^{K_u} r_{jku}^2}}, \quad d_{iju} = 1 - O_{iju}. \quad (1)$$

The index is defined by

$$\text{FDI}_u = \underbrace{\left(1 - \sum_{i=1}^{n_u} p_{iu}^2\right)}_{E_u} \underbrace{\left[\frac{1}{2} \sum_{i=1}^{n_u} \sum_{j=1}^{n_u} p_{iu} p_{ju} (1 - O_{iju})\right]}_{D_u} = E_u D_u. \quad (2)$$

The first factor E_u has the Gini-Simpson form but is applied to consumption shares rather than species abundances; D_u is a consumption-weighted, Rao-style dietary dissimilarity component. The factor $1/2$ counts each unordered pair once because $d_{iju} = d_{jiu}$ and $d_{iiu} = 0$; it is retained from the original simulation rather than replaced by a different convention for Rao's quadratic entropy. The functional form is invariant to a common multiplicative change in all c_{iu} within the same community, but not generally invariant to changes in consumer membership or aggregation.

A simple interpretation helps explain what the product rewards. If nearly all consumption is concentrated in one consumer, E_u is small even when the remaining consumers have very different diets. If consumption is spread more evenly but the consumers rely on nearly identical resources, D_u is small. FDI is relatively high only when consumption is distributed across consumers and their diets are differentiated. It is a description of those two conditions together, not a direct measurement of productivity, ecosystem health, or the number of ecological functions present.

The original coauthored preprint [22] used simulated organism-level oxygen consumption. The present Jinsha and Coweeta applications instead use estimated food-consumption fluxes. Oxygen use and food intake are not physically interchangeable, even though each supplies normalized weights in the same equation. The present article does not repeat the original NetLogo figures or statistical simulation results. An exact factorization clarifies the interpretation of the index: with $E_u > 0$, define the mean dietary dissimilarity of two consumers sampled in proportion to their consumption weights, conditional on their being different consumers,

$$\bar{d}_{w,u} = \frac{\sum_{i \neq j} p_{iu} p_{ju} (1 - O_{iju})}{E_u}, \quad D_u = \frac{E_u}{2} \bar{d}_{w,u}, \quad \boxed{\text{FDI}_u = \frac{E_u^2}{2} \bar{d}_{w,u}}. \quad (3)$$

This relationship shows that consumption evenness enters the product twice: first through pairwise consumer sampling in D_u , and again as the external multiplier E_u . The additional weighting of evenness is an interpretable design choice, not independently proven ecological superiority. A complete evaluation should compare FDI_u to E_u , D_u , and the conditional weighted mean $\bar{d}_{w,u}$ rather than treating the product as mathematically independent of its factors. In particular, D_u already contains the probability of drawing two different consumers under consumption-weighted sampling. Multiplication by E_u makes the final score especially sensitive to whether consumption is shared broadly among consumers. That emphasis may be useful when the research question concerns the joint balance of participation and dietary separation, but it can also make the score less responsive to niche differentiation when one consumer dominates. The present analyses examine this trade-off rather than assuming that additional weighting is always desirable.

The number of consumers also affects the scale of the calculation. For n_u consumers with positive consumption, $E_u \leq 1 - 1/n_u$;

because dietary dissimilarity lies between zero and one, the conditional form gives

$$0 \leq \text{FDI}_u \leq \frac{1}{2} \left(1 - \frac{1}{n_u} \right)^2. \quad (4)$$

This is a mathematical ceiling, not a claim that a community with a limited number of resource categories can attain it. More importantly, FDI values calculated for six aggregated Jinsha consumer groups and for eleven to fifteen stream taxa should not be compared as if they represented the same sampling resolution. Within each analysis, reporting the consumer count, component values, and definition of food consumption makes the index more informative than an isolated score.

4 Temporal Food-Web Assessment in the Lower Jinsha River

4.1 Rationale and scope

The previously reported NetLogo simulation [22] specified organism-level consumption proportions using oxygen consumption and generated dietary proportions from randomized ecological assignments. An initial empirical assessment must separate the mathematical framework from the measurement system used to populate it. A publicly released dataset describing hydrodynamic regulation and trophic dynamics downstream of Xiangjiaba Dam in the lower Jinsha River provides a first test of the framework's computational transferability. The deposited collection contains 14 CSV files covering biological observations, stable isotopes, MixSIAR source estimates, Ecopath model inputs, hydrodynamic descriptors and water quality. The analysis reported here uses only quantities actually evaluated in the current four-period calculation; the remaining data are context and opportunities for further investigation rather than unreported validation experiments. The deposit and its documentation were published by Ren and colleagues (2026) [17].

This extension is a preliminary empirical application, not a fully independent validation. The availability of measured fish counts and stable isotopes is valuable, but the group-level diet matrices and consumption-to-biomass parameters used to construct FDI are model-derived. Accordingly, the word “external” refers to the use of data from outside the original NetLogo simulation, not to complete statistical independence among all modeled ecosystem quantities.

4.2 Study area, sampling periods and analytical unit

The source study covers an approximately 14-km reach downstream of Xiangjiaba Dam in China. The source archive distinguishes May 2022 (pre-flood), October 2022 (post-flood), May 2023 (pre-flood), and June 2023 (pre-flood following an antecedent short-term rise–fall hydrodynamic event). The environmental sampling sections are L1, H1, R1, and R2. The available Ecopath parameter files, dietary matrices, and isotopic niche metrics have one aggregate result per period, not one independent result per sampling section. Consequently, the inferential unit for the reported FDI correlations is **the sampling period**, and $n = 4$, not the number of fish individuals, site records, food-web rows, or measured isotope samples.

Table 1: Source periods, dataset labels, and hydrological context from the Dryad metadata.

Sampling period	Ecopath notation	Diet-matrix notation	Reported hydrological phase
May 2022	22-May	May-22	Pre-flood
October 2022	22-Oct	Oct-22	Post-flood
May 2023	23-May	May-23	Pre-flood
June 2023	23-Jun	Jun-23	Pre-flood; preceding short-term rise–fall event

4.3 Variables and operational definition

The empirical construction retains the multiplicative structure of the original index and its one-half multiplier. Its input is modeled annual food-consumption flux, not directly measured respiration or oxygen uptake. For each consumer group i in period t , estimated consumption is

$$Q_{it} = B_{it} \left(\frac{Q}{B} \right)_{it}, \quad p_{it} = \frac{Q_{it}}{\sum_{j=1}^n Q_{jt}}, \quad n = 6, \quad (5)$$

where B is Ecopath group biomass (t km^{-2}), (Q/B) is the corresponding annual consumption/biomass ratio (year^{-1}), and Q thus has the dimension $\text{t km}^{-2} \text{ year}^{-1}$. The proportions p_{it} sum to unity within a period. The six consumers are herbivorous fish, omnivorous fish, filter-feeding fish, carnivorous fish, benthos, and zooplankton; their nine-component dietary vectors are obtained from the period-specific Ecopath prey-by-consumer matrix. Groups with undefined Q/B (the primary-production and detrital input groups) are not treated as consumers in the weighting calculation.

Let r_{ikt} be the proportion of resource k in the diet of consumer i during period t . The Pianka overlap and dietary distance are

$$O_{ijt} = \frac{\sum_{k=1}^9 r_{ikt} r_{jkt}}{\sqrt{\sum_{k=1}^9 r_{ikt}^2} \sqrt{\sum_{k=1}^9 r_{jkt}^2}}, \quad d_{ijt} = 1 - O_{ijt}. \quad (6)$$

The empirical index follows the original two-factor construction:

$$E_t = 1 - \sum_{i=1}^6 p_{it}^2, \quad (7)$$

$$D_t = \frac{1}{2} \sum_{i=1}^6 \sum_{j=1}^6 p_{it} p_{jt} d_{ijt}, \quad (8)$$

$$\text{FDI}_t = E_t D_t. \quad (9)$$

The factor 1/2 counts unordered consumer pairs once. In some ecological conventions, “Rao’s quadratic entropy” is written as the full double sum without this factor; D_t is therefore called the Rao-style dietary dissimilarity component here, without changing the original equation. The component E_t is a Gini–Simpson-form evenness/concentration complement applied to consumption weights. It should not be confused with the Gini–Simpson index calculated separately from fish species counts. Neither Q_{it} nor ambient dissolved-oxygen concentration is an observed organism-specific oxygen-consumption rate, and neither is a direct observation of GPP.

4.4 Data cleaning, alignment, and quality control

The original data archive uses three synonymous date formats. Periods were matched explicitly rather than by lexicographic sorting (Table 1). Functional-group labels were converted to lower case and hyphens were replaced with spaces to align the “Filter-feeding fish” entry in the Ecopath parameter table with the “filter feeding fish” column in the diet matrix. Diet proportions were checked to sum to one for all 24 consumer-period combinations. In every period, six weighted consumer labels were matched to six consumer diet columns. The index was independently recomputed using explicit name-based ordering, with results agreeing with the first implementation in all four periods. The alternative biomass weights were also verified to sum to one within period.

The June 2023 source documentation states that filter-feeding fishes were not collected in that field survey, although the functional group was retained in the June 2023 Ecopath representation. The analysis retains the group exactly as in the published model and does not equate the model’s retained group with a positive June survey catch. This is an important qualification for ecological interpretation.

4.5 Comparison variables and analysis plan

The fish survey records provide species-count-based richness, Shannon entropy ($H' = -\sum_s a_s \ln a_s$), and the Gini–Simpson index ($1 - \sum_s a_s^2$), where a_s denotes the relative count share of taxon s within a survey period. Species records were associated with one of the four periods using the archive’s date crosswalk. Published period-level corrected standard ellipse area (SEAc) provides a descriptive isotopic niche-breadth comparator; it is a measure of isotopic dispersion, not a Pianka overlap coefficient. Published total ecosystem conversion efficiency provides a separate Ecopath-derived descriptor, but shares a food-web modeling framework with the dietary inputs to FDI.

Three checks were applied: (i) reproduction of the original mathematical calculation with consistent group ordering; (ii) comparison of the index and its separate E_t and D_t factors against species and isotopic descriptors, including descriptive Pearson correlations; and (iii) sensitivity to a change from modeled consumption weights to simple biomass weights while holding the diet vectors and functional form fixed. All correlations use $n = 4$ period-level observations. Owing to the tiny, nonindependent temporal sample, these correlations are descriptions only: no null-hypothesis significance, predictive out-of-sample claims, or causal interpretations are assigned.

4.6 Primary empirical results

4.6.1 Complete period-level comparison

Table 2: Four-period empirical results. FDI, consumption evenness E_t , and dietary dissimilarity D_t are calculated from Ecopath-based inputs; Shannon and Gini–Simpson are derived from fish counts; SEAc and ecosystem conversion efficiency are period-level source descriptors.

Period	Richness	Shannon	Gini–Simpson	E_t	D_t	FDI_t	SEAc	Ecopath eff.
May 2022	24	2.9272	0.9354	0.605010	0.108134	0.065422	17.01	9.731
Oct. 2022	20	2.9344	0.9439	0.538064	0.110640	0.059531	8.22	10.040
May 2023	29	3.2539	0.9572	0.606127	0.091007	0.055162	21.54	9.185
June 2023	16	2.6611	0.9238	0.603347	0.094067	0.056755	16.21	10.770

Consumption-weighted FDI ranged from 0.055162 (May 2023) to 0.065422 (May 2022), an 18.6% difference relative to the lowest observed value. Its ordered ranking was May 2022 > October 2022 > June 2023 > May 2023. Dietary dissimilarity ranged from 0.091007 to 0.110640; consumption evenness ranged from 0.538064 to 0.606127. The source values are shown in Table 2 and Figure 1. The simulation and these empirical values should not be placed on a common absolute scale because they differ in unit

of analysis, number of consumers, and meaning of the consumption variable; simulation results are available in the cited preprint [22].

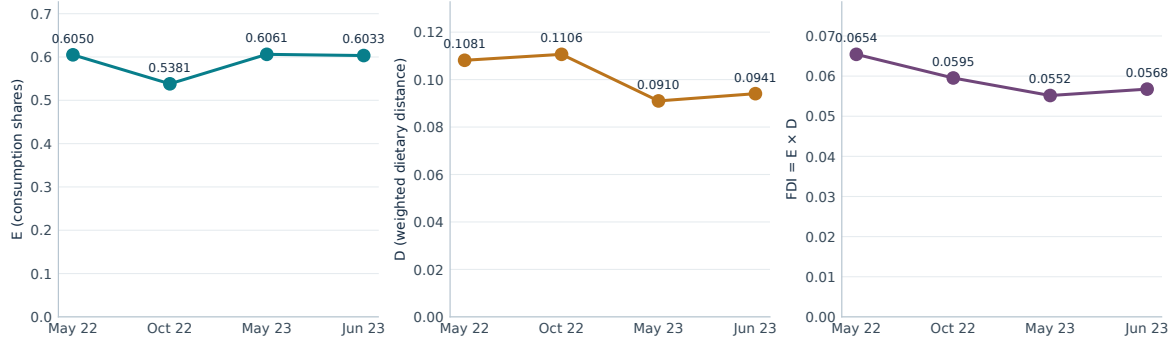


Figure 1: FDI and its two factors across four categorical sampling periods. Connecting lines show observed order only; they are not interpolated trajectories between irregularly spaced samples.

4.6.2 Rank divergence from taxonomic and isotopic diversity

The most notable contrast occurs in May 2023: fish richness (29), Shannon diversity (3.2539), Gini–Simpson diversity (0.9572), corrected isotope ellipse area (21.54), and consumption evenness (0.606127) are all at their four-period maxima, whereas dietary dissimilarity (0.091007) and integrated FDI (0.055162) reach their four-period minima. The ranking illustrates that a large species-diversity score and broad isotopic niche space do not necessarily coincide with high consumption-weighted dietary separation under this specific aggregation. It does not establish ecological deterioration in May 2023; the metrics operationalize different quantities and have no independent ecological “health” ground truth in this analysis.

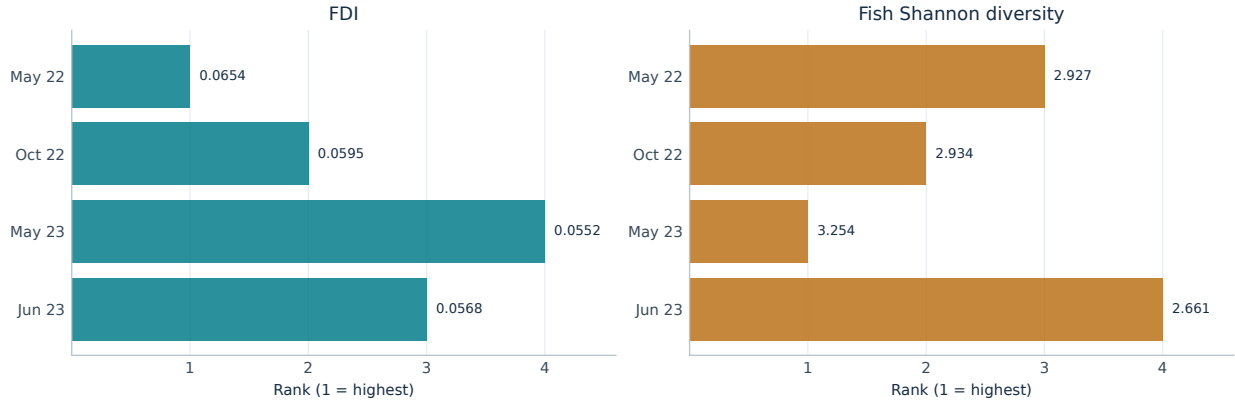


Figure 2: Rank-based descriptive comparison. A lower numerical rank means a higher value for that particular metric. Rank differences demonstrate distinct behavior but not superiority or statistical independence.

4.6.3 Contribution of each component and connection with Rao-style dissimilarity

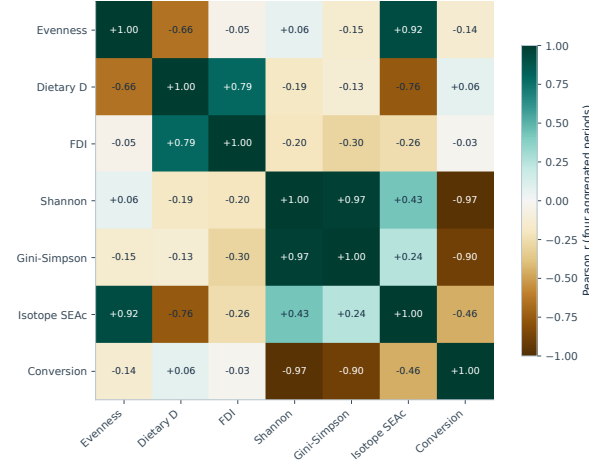
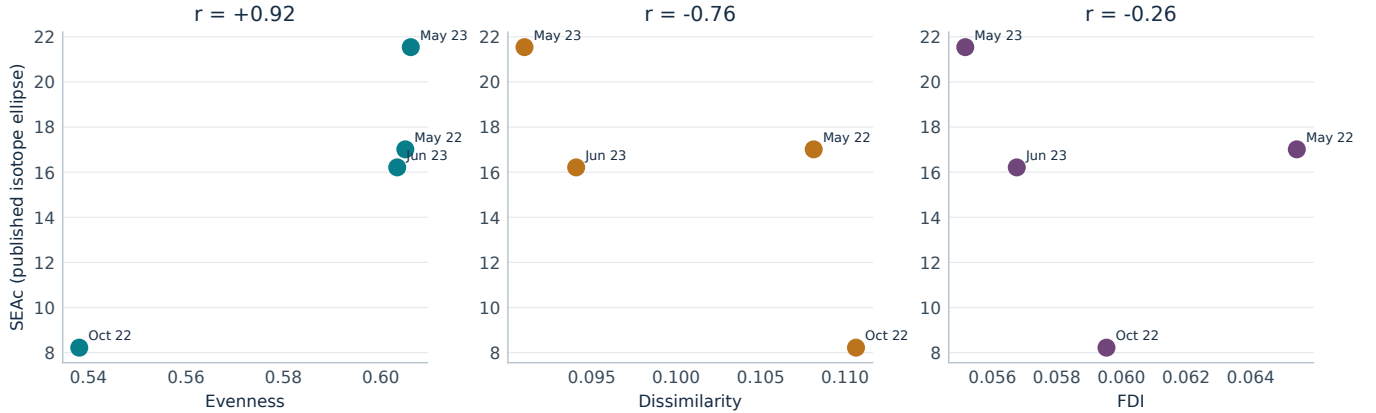
The index preserves the identity $FDI_t = E_t D_t$. Although the FDI ranking is not identical to the ranking of E_t or D_t , changing ranks are an expected algebraic consequence of multiplication and cannot alone establish predictive value. In October 2022, dietary dissimilarity is largest ($D_t = 0.110640$), but consumption evenness is smallest ($E_t = 0.538064$); the resulting FDI is intermediate rather than maximal. In May 2023, the greatest evenness coincides with the least dietary dissimilarity, producing the lowest integrated FDI. Pearson correlation between FDI and dietary dissimilarity is $r = 0.786$, whereas correlation between FDI and evenness is $r = -0.047$ across the four periods. The latter is possible because the two input factors covary negatively and the analysis spans very few observations. This does not make dietary dissimilarity a causally identified driver, nor does it show that the product outperforms D_t by itself.

4.6.4 Descriptive correlation structure

The four-period component comparison yielded $r(E_t, SEAc_t) = +0.918$, $r(D_t, SEAc_t) = -0.761$, and $r(FDI_t, SEAc_t) = -0.256$. In this dataset, the composite FDI does not show a stronger linear association with isotopic niche breadth than its ingredients. An apparently large r based on four observations is unstable, and SEAc is not a definitive measurement target for FDI. Likewise, Ecopath-derived conversion efficiency is not an independent ecological criterion when interpreting model-derived diets. Figures 3 and 4 display these descriptive relations without fitted significance claims.

Table 3: Exploratory Pearson correlations ($n = 4$). These are descriptive associations and should not be interpreted as stable population parameters or significance tests.

Metric	With FDI_t	With E_t	With D_t	With SEAc
Consumption evenness (E_t)	-0.047	1.000	-0.655	0.918
Dietary dissimilarity (D_t)	0.786	-0.655	1.000	-0.761
FDI	1.000	-0.047	0.786	-0.256
Shannon index	-0.203	0.060	-0.188	0.426
Gini-Simpson index	-0.300	-0.149	-0.132	0.244
Ecopath conversion efficiency	-0.028	-0.143	0.065	-0.456

**Figure 3:** Pearson correlation matrix across four aggregate period observations. The numerical values are exploratory; shared model inputs and $n = 4$ constrain inference.**Figure 4:** Association of consumption evenness, dietary dissimilarity, and integrated FDI with SEAc. Labels denote periods rather than independent sampling stations. No regression lines or confidence intervals are shown because only four observations are available.

4.6.5 Sensitivity to the representation of consumer contribution

A second index was computed using $\tilde{p}_{it} = B_{it}/\sum_j B_{jt}$ in place of $p_{it} = Q_{it}/\sum_j Q_{jt}$ in both the evenness and dietary dissimilarity factors. Dietary profiles, consumer groups, and the original factor $1/2$ were unchanged. This is a coherent alternative operationalization, not a perturbation of the final numerical FDI values. Results appear in Table 4 and Figure 5. The highest-ranked

Table 4: FDI sensitivity to consumer weights. Values are matched to the same period-specific dietary matrices.

Period	Consumption-weighted	Biomass-weighted	Consumption rank	Biomass rank
May 2022	0.065422	0.049944	1	3
Oct. 2022	0.059531	0.052065	2	2
May 2023	0.055162	0.046142	4	4
June 2023	0.056755	0.064311	3	1

period changes from May 2022 under consumption weighting to June 2023 under biomass weighting, although May 2023 remains lowest under both. The two four-point index series are weakly negatively correlated ($r = -0.181$), emphasizing that the particular proxy for consumer contribution can alter both numerical magnitudes and temporal rankings. This constitutes evidence of sensitivity, not evidence that either weighting method is inherently more ecologically accurate. The consumption-weighted version is retained as the primary adaptation because it more closely approximates the intent of the original consumption-based construction,

while the biomass-weighted version is treated as a robustness diagnostic.

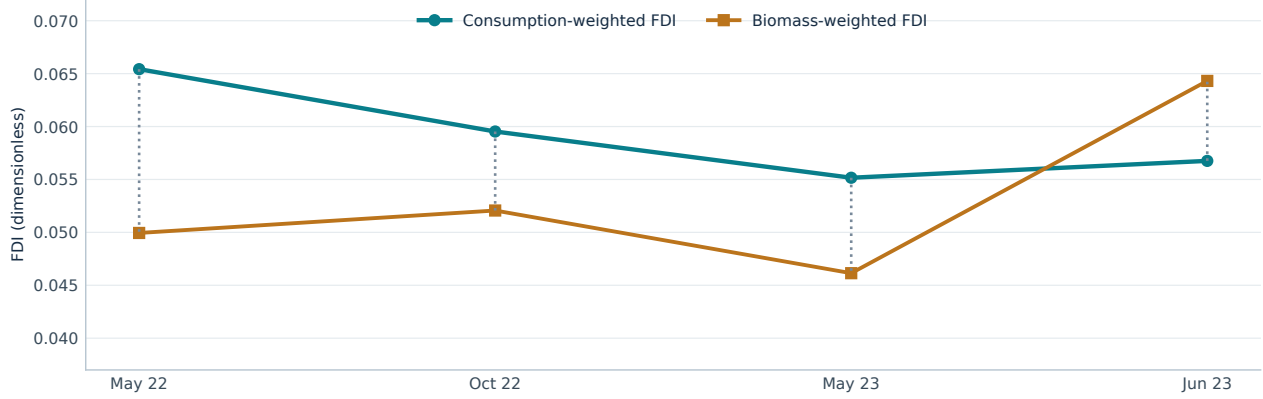


Figure 5: Direct comparison of consumption- and biomass-weighted FDI. The consumer diets are unchanged; only the period-level shares p_{it} used in both terms of the index differ.

4.6.6 Exact descriptive decomposition of period-to-period FDI changes

Since $FDI_t = E_t D_t$, the change between any two successive sampled periods satisfies

$$\ln\left(\frac{FDI_{t+1}}{FDI_t}\right) = \ln\left(\frac{E_{t+1}}{E_t}\right) + \ln\left(\frac{D_{t+1}}{D_t}\right). \quad (10)$$

This is an exact accounting identity, not a causal decomposition or an independent statistical fit. It describes how the two calculated components jointly give the observed product changes without needing additional raw data. The May–October 2022 transition shows an 11.1% decrease in evenness and a 2.3% increase in dietary dissimilarity, producing an approximately 9.0% decline in FDI. From October 2022 to May 2023, evenness increases 12.6% while dissimilarity decreases 17.7%, yielding a 7.3% FDI decline. From May to June 2023, evenness decreases 0.46% and dissimilarity increases 3.36%, yielding a 2.89% increase in FDI. Figure 6 shows the additive log terms. The October 2022–May 2023 interval spans distinct seasons and years; none of these comparisons isolates dam regulation or any specific mechanism.

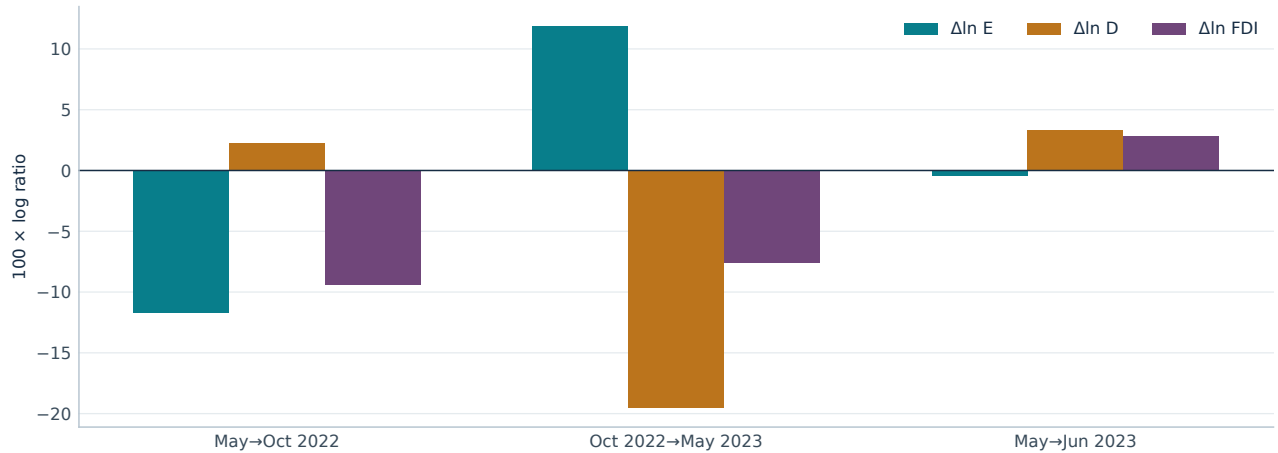


Figure 6: Exact accounting decomposition $\Delta \ln FDI_t = \Delta \ln E_t + \Delta \ln D_t$ for successive sampled periods. This is a descriptive algebraic identity, not a causal or controlled treatment-effect analysis.

4.7 Interpretation and evidential limits

The Jinsha assessment shows that the complete mathematical framework can be transferred to an external food-web representation when the consumption variable, taxonomic aggregation, and dietary-resource matrix are explicitly defined. The difference between the Shannon and FDI rankings is particularly informative because the two measures answer different questions. Shannon diversity describes how fish abundance is distributed among species. FDI, calculated here from six modeled consumer groups, describes how estimated food intake is distributed among those groups and how different their modeled diets are. A community may therefore be taxonomically diverse without having a high value for this particular consumption-weighted combination. This is a reason to examine the two measures together, not a reason to discard either one. Its rankings differ from those of fish richness, Shannon diversity, and corrected isotopic ellipse area, and replacing consumption weights with biomass weights changes the highest-ranked period. These differences indicate that the weighting and niche inputs carry information not encoded in a species-count distribution

alone. They do not demonstrate that the product predicts an independently measured ecological outcome more accurately than its individual components or established biodiversity indices; in this four-period dataset, the FDI–SEAc correlation is weaker than the corresponding correlations of either E_t or D_t with SEAc.

A central limitation is the change in measurement and resolution between the original simulation and this field application. Simulated units were individual organisms with oxygen-consumption assignments, whereas the field calculation uses six Ecopath groups, modeled food intake $Q_{it} = B_{it}(Q/B)_{it}$, and modeled diets. Ecopath conversion efficiency is not fully independent of related food-web estimates. Moreover, the June 2023 model retains a filter-feeding group that was not collected in that month’s survey. The four time periods are not replicated experimental treatments, and their small number precludes reliable estimation of population correlations or response prediction. Sensitivity to alternate consumer weights therefore remains an essential qualification of the descriptive findings.

5 Dietary Differentiation Across Singapore Reservoirs

5.1 Research purpose and source data

The FDI proposed in the earlier preprint [22] was designed to represent two ecological dimensions together: how the total consumption activity is distributed among organisms and how different their resource-use patterns are. A second empirical dataset from Wilkinson et al. (2022) [18,19] provides a spatial assessment of the latter dimension across 12 tropical Singapore reservoirs, classified in the published site-information file as urban (3), estuarine (5), or forest (4). The public archive contains 12 resource-by-consumer predation matrices, fish gut-content compositions, a fish variable dictionary, taxon codes, and reservoir descriptors. The matrices describe estimated relative trophic link strengths, informed by gut-content and isotopic evidence; they do not give oxygen-consumption rates or absolute group-level food-consumption flux. The Singapore analyses therefore characterize Pianka-based dietary differentiation and the importance of community coverage rather than independently validating numerical FDI values.

5.2 Reproducible data preparation and quality control

The reservoir metadata required `fileEncoding="latin1"` to import non-ASCII site-text fields, with some degree symbols and date separators still imperfectly rendered; numeric and categorical fields used here were preserved. The fish dictionary documented 18 variables, including 12 gut-content categories. Fish diets were read from `diet_composition_fish.csv`: 103 rows, each a unique reservoir–fish-species combination, with 4–17 fish species per reservoir. The category `substrate` represents rock/sand rather than a dietary food source and was excluded from niche comparisons. Among 38 nonmissing substrate values, the mean substrate share was 0.0447 and the largest was 0.233. After checking that each original row totaled 0.998–1.002, empty dietary categories were provisionally treated as zero, substrate was excluded, and the remaining 11 categories were divided by each record’s positive food total. All 103 resulting 11-resource vectors summed to one and contained no numeric missing values. These NA-to-zero decisions are conditional on the compositional reporting convention, rather than proof of true biological absences.

The 12 separate predation matrices were each imported with `check.names=FALSE`. Their first column identifies resources and their remaining columns identify consumers; entries marked – were treated as absent feeding links (zero). The matrices contained 22–37 named ecological groups, including 18–30 consumers with positive normalized diet columns and 3–7 basal zero-sum columns. All 12 matrices had no unparseable entries, and all positive consumer columns summed to one within 10^{-3} . Basal zero-sum columns were excluded from the overlap calculation, while their resource rows were retained.

5.3 Dietary niche methods and inferential unit

For each reservoir, the fish-only matrix contained one row per observed fish species and 11 normalized diet categories. Pianka overlap between species i, j was calculated using Equation (1), and mean fish-only dissimilarity was computed as

$$\bar{d}_r = \frac{2}{m_r(m_r - 1)} \sum_{1 \leq i < j \leq m_r} (1 - O_{ijr}), \quad (11)$$

where m_r is the number of fish species sampled in reservoir r . The same unweighted-pair method was applied to all positive consumer columns in each whole-food-web matrix. Both results assign equal weight to distinct pairs, and neither should be confused with the consumption-weighted D_u factor of FDI. The statistical unit for reservoir-type inference was the reservoir ($n = 12$), not an individual pair of fish or food-web consumers.

Reservoir-level comparisons included group means and SDs, Kruskal–Wallis rank tests, a 9,999-label-permutation Kruskal–Wallis test for the fish-only data (seed 2026; group sizes preserved), and Spearman correlations with observed fish-species count and between fish-only and whole-food-web means. To assess sensitivity to coverage, four fish species per reservoir were randomly selected 1,000 times (seed 2026); the resulting mean dissimilarities were summarized by reservoir. A further 1,000 draws from these replicate vectors assessed the forest–urban difference. The resulting subset quantiles are not sampling confidence intervals, and this exercise does not standardize species identities between communities. All statistical comparisons are exploratory, with small, unequal group sizes and no causal assignment of reservoir type.

5.4 Results: fish-only and whole-food-web structure

Table 5: Singapore dietary niche differentiation. Mean $\bar{d}_r$ is the unweighted pairwise complement of Pianka overlap, not a consumption-weighted FDI.

Reservoir type	Sites	Fish $\bar{d}_r$	Fish SD	Food-web $\bar{d}_r$	Food-web SD
Urban	3	0.5291	0.0514	0.7341	0.0285
Estuarine	5	0.6399	0.0738	0.6887	0.1069
Forest	4	0.6980	0.0717	0.7292	0.0702

The mean fish-only dissimilarity ranged from 0.4956 (Bedok, Res 1) to 0.7743 (MacRitchie, Res 9), compared with 0.5643 (Serangoon, Res 5) to 0.8223 (Kranji, Res 12) in the broader food webs. The reservoir-type pattern differed by community scope (Table 5; Figures 7–8). In fish-only data, the forest mean (0.6980) exceeded the estuarine (0.6399) and urban (0.5291) means; the Kruskal–Wallis test gave $H = 6.0256$, $df = 2$, $p = 0.04915$, with the label-permutation test giving $p = 0.0366$. Fish-species count was weakly associated with fish-only $\bar{d}_r$ (Spearman $\rho = -0.110$, $p = 0.7347$), although low power and variation in species identity remain substantial concerns.

Species-count standardization gave a positive forest–urban difference in 97.2% of 1,000 simulated four-species comparisons, with mean difference 0.1621 and central 95% subset range $[-0.0035, 0.3674]$. Because that range includes zero and random subsampling preserves the original unweighted pairwise mean in expectation, these results describe species-selection sensitivity rather than independent verification of a forest effect.

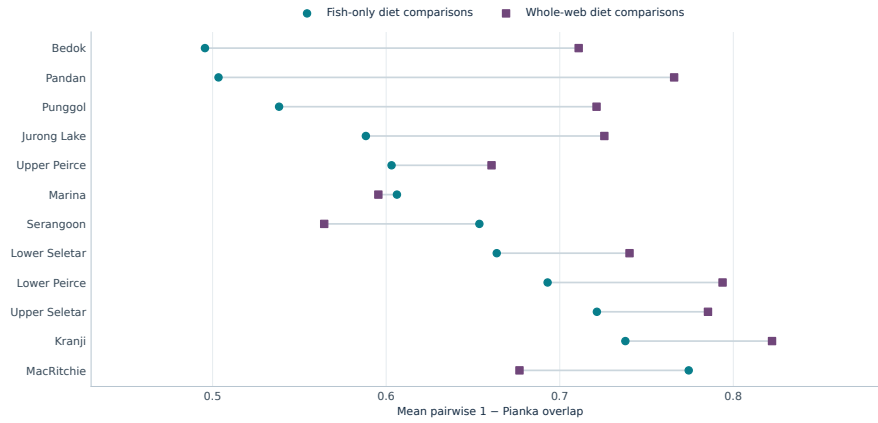


Figure 7: Fish-only versus whole-food-web mean dietary dissimilarity for each of the 12 reservoirs. Lines connect two differently scoped estimates at the same reservoir; they are not time trends. Values were plotted from the recorded rounded reservoir-level result table.

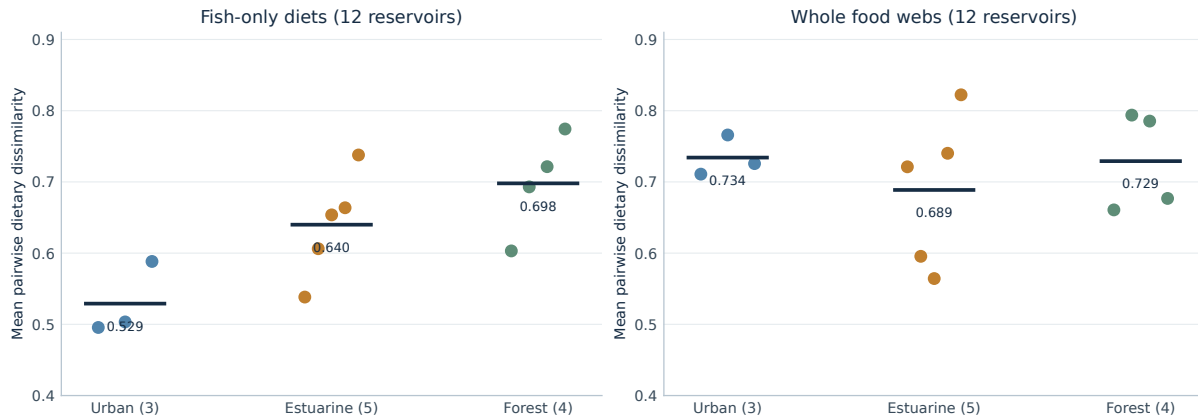


Figure 8: Dietary dissimilarity across urban, estuarine, and forest reservoir categories. Each point represents one reservoir, and horizontal bars indicate category means. The paired panels compare fish-only and whole-food-web dietary dissimilarity. Small category sample sizes limit statistical inference.

In the full food webs, urban (0.7341) and forest (0.7292) average dissimilarities were nearly identical, and the reservoir-type test was not statistically significant ($H = 0.3231$, $df = 2$, $p = 0.8508$). Fish-only and whole-web reservoir rankings were weakly associated (Spearman $\rho = 0.2727$, $p = 0.3911$). The disagreement does not itself constitute a statistically tested interaction between scope and reservoir type, nor can it be called a causal environmental response: fish-only profiles and food-web profiles include different organisms, different resources, and partly related original evidence. Their values should be understood as different ecological representations, not as estimates of one invariant quantity.

5.5 Interpretation: Dietary structure depends on community scope

The Singapore case demonstrates that Pianka-based dissimilarity depends on the specified set of consumers and dietary resources. The contrast between the fish-only and whole-food-web analyses is useful because it reveals a boundary condition for interpretation. A reservoir may contain fish with relatively distinct diets while its wider set of consumers shows a different pattern once invertebrates and other feeding links are included. A fish-only reservoir pattern did not persist when broader food-web consumers and resources were included. Because the two representations differ in both community membership and dietary resolution, the comparison does not identify a causal effect of reservoir type or a formal statistical interaction between reservoir type and food-web scope. It does, however, show that a niche-only characterization of fish diets need not describe the wider trophic community.

The available dietary fractions cannot supply the consumer-level total intake shares p_{ir} needed for full FDI. Summing a normalized dietary vector cannot recover how much that consumer ate. If equal consumer shares were hypothetically imposed within reservoir r ($p_{ir} = 1/m_r$), Equation (2) would simplify to

$$\text{FDI}_{r,\text{equal}} = \frac{1}{2} \left(1 - \frac{1}{m_r} \right)^2 \bar{d}_r. \quad (12)$$

For a fixed consumer count, this is only a rescaled form of the unweighted mean dietary dissimilarity, not evidence for the contribution of measured consumption weights. Full FDI is therefore intentionally not assigned to these reservoirs. The complementary contribution of this dataset is a reproducible assessment of the dietary term and a demonstration of sensitivity to ecological coverage.

6 Consumption Structure in Experimental Coweeta Streams

6.1 Study design, data provenance, and sample-unit definition

A third public dataset supplies the missing information needed to evaluate the complete consumption-weighted FDI: resource-specific amounts of food consumed by identifiable macroinvertebrate taxa. Demi et al. (2020) examined five headwater streams (6–10) at the Coweeta Hydrologic Laboratory, North Carolina, following one pretreatment year (PRE: July 2010–June 2011) and two years of experimental nitrogen and phosphorus enrichment (YR1: July 2011–June 2012; YR2: July 2012–June 2013) [20,21]. Streams received different nutrient concentrations and target N:P ratios (2:1–128:1), rather than a single identical dose. Source files are deposited at <https://doi.org/10.5061/dryad.n02v6wwt4>.

The input is the TBoFP and Material Flows by Tax worksheet in `Trophic_Basis_of_Production_and_Material_Flows.xlsx`, which contains stream, study year, taxon, resource category, and estimated consumption. The field Consumed is expressed as g ash-free dry mass $\text{m}^{-2} \text{year}^{-1}$; it is not direct oxygen consumption, productivity, or biomass. The data combines gut-content information and secondary-production calculations, so intake should be described as estimated. The independent statistical unit for longitudinal comparisons is the stream ($n = 5$), with three repeated periods each ($n = 15$ observed stream-years), not the number of spreadsheet rows or taxonomic resource links.

6.2 Corrected cleaning, aggregation rules, and index calculation

The original worksheet contains individual consumer taxon records and spreadsheet subtotal rows labeled Total and Predators. These subtotal records were excluded to prevent double counting of resource flows. Data preparation excluded 105 aggregate rows, four blank rows, and 12 rows with unreported consumption values belonging to two unusable taxon profiles. The corrected analytical matrix contains 1,152 resource observations, 192 positive-intake consumer profiles, and 15 stream-year samples. Unreported consumer-resource cells are set to zero only after the missing-data exclusions, with reported zeros retained. The 11 variant resource labels are consolidated into six categories: amorphous detritus, animals, diatoms, fungi, leaves, and wood.

For taxon i , stream s , period t , and resource k , estimated resource intake C_{iskt} provides

$$C_{ist} = \sum_{k=1}^6 C_{iskt}, \quad p_{ist} = \frac{C_{ist}}{\sum_j C_{jst}}, \quad r_{iskt} = \frac{C_{iskt}}{C_{ist}}. \quad (13)$$

The six-component dietary vector is normalized within each taxon, and consumer weights are normalized within each stream-year. Applying Equations (1) and (2) gives E_{st} , D_{st} , and FDI_{st} . The same algebraic definition is used for all three years; the number of identifiable consumer taxa is allowed to differ by year, as shown in Table 6. Importantly, the full index is recalculated after a shared-taxon restriction; earlier full-sample consumer weights are not carried over.

6.3 Complete results: consumption volume, evenness, and dietary separation

In both enrichment years, total estimated consumption exceeded pretreatment in every stream. The first-year consumption increases ranged from 18.4% (Stream 7) to 78.8% (Stream 6); second-year increases ranged from 12.8% (Stream 6) to 180.9% (Stream 10). Despite this consistent increase in total throughput, the corrected index increased in only one stream (Stream 8) in each enrichment year. The key ecological implication is not that increasing total flow was beneficial or harmful for diversity, but that total consumption and its distribution among consumers with different diets answer different questions.

Table 6: Corrected 15 stream–year calculations. n is retained consumer taxa, m is taxa found in every period within a stream, C is estimated consumption (g AFDM m^{-2} year $^{-1}$), and matched FDI recalculates both components after restricting to those m taxa. All index values are dimensionless.

Stream	Period	n	m	E_{st}	D_{st}	FDI_{st}	C_{st}	$FDI_{st}^{matched}$
6	PRE	13	9	0.7992	0.0312	0.02490	87.85	0.01372
6	YR1	13	9	0.6686	0.0314	0.02103	157.05	0.01623
6	YR2	13	9	0.7832	0.0314	0.02459	99.05	0.02058
7	PRE	13	10	0.6880	0.0220	0.01516	109.61	0.01391
7	YR1	11	10	0.7430	0.0130	0.00965	129.80	0.00890
7	YR2	12	10	0.8214	0.0161	0.01322	130.39	0.01263
8	PRE	12	10	0.7316	0.0141	0.01034	83.14	0.00952
8	YR1	12	10	0.7092	0.0720	0.05107	116.45	0.03452
8	YR2	12	10	0.7444	0.0497	0.03700	133.72	0.03578
9	PRE	12	11	0.8069	0.0726	0.05857	116.00	0.05784
9	YR1	15	11	0.7170	0.0427	0.03063	154.22	0.02864
9	YR2	14	11	0.7711	0.0424	0.03273	180.33	0.03222
10	PRE	13	10	0.7538	0.0384	0.02895	87.91	0.02768
10	YR1	14	10	0.7488	0.0357	0.02676	124.29	0.02557
10	YR2	13	10	0.4312	0.0299	0.01290	246.97	0.01113

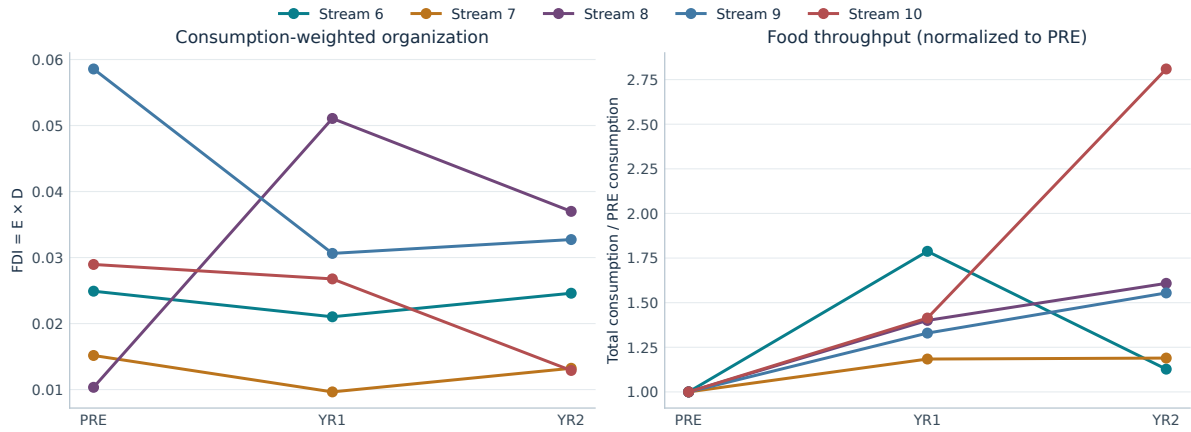


Figure 9: FDI and total resource throughput answer different questions. Lines connect repeated measurements from the same five streams; the throughput panel divides each period's total estimated intake by that stream's pretreatment value.

Stream 8 provides a clear within-community illustration: with 12 retained taxa in all three periods, its FDI rose from 0.01034 in PRE to 0.05107 in YR1 and 0.03700 in YR2, while D increased from 0.01413 to 0.07201 and 0.04970. The first-year increase was therefore predominantly associated with consumption-weighted dietary differentiation rather than a comparable increase in evenness. Stream 10, by contrast, increased consumption by 180.9% in YR2 while FDI declined from 0.02895 to 0.01290 (approximately -55.5%); evenness decreased from 0.75379 to 0.43119 and weighted dissimilarity from 0.03841 to 0.02991. None of these observations independently establishes better or worse ecosystem condition.

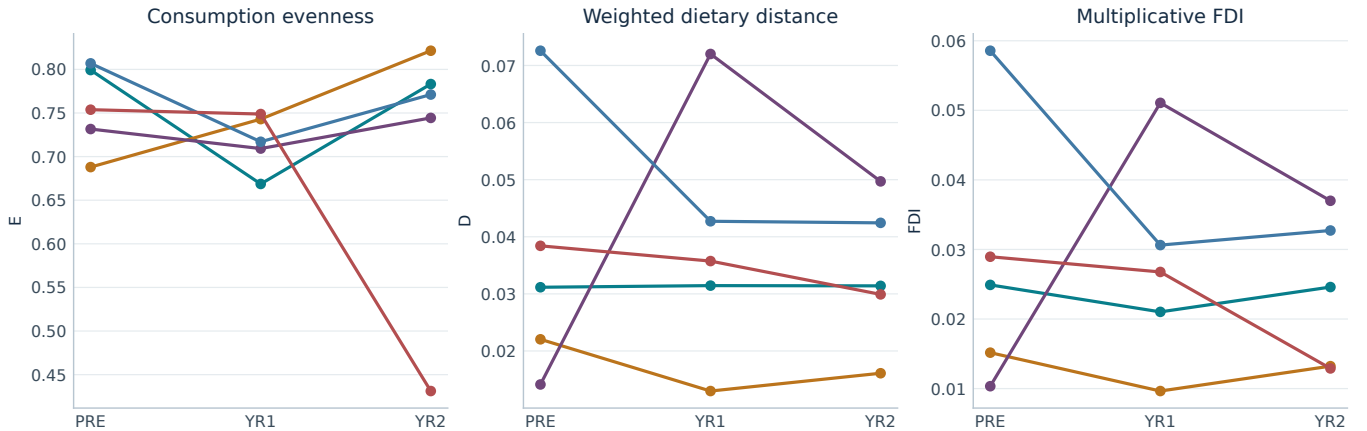


Figure 10: Every corrected stream–year factor, shown separately. The exact $E \times D$ relationship must be considered when interpreting the product; these panels represent related mathematical summaries, not three independently measured ecological outcomes.

6.4 Incremental effect of the outer evenness term

The stream data also allow a direct comparison of FDI with its Rao-style dietary component D , which is particularly important because the two quantities are constructed from the same diet and consumption estimates. Across the 15 stream–years, their rankings were strongly associated (Spearman $\rho = 0.971$). Seven observations changed ordinal positions after multiplication by

evenness, but the largest change was only two places. The product therefore offers an explicit reweighting of an existing dietary summary rather than an independent source of information.

Stream 6 shows where that reweighting matters. Its D increased slightly from 0.031160 before enrichment to 0.031446 in the first year and 0.031403 in the second, while FDI decreased from 0.024904 to 0.021026 and 0.024594, respectively. Declining or incompletely recovered consumption evenness outweighed the small increase in weighted dietary separation. This is the clearest observed consequence of the outer multiplier: a change can be positive for D yet negative for FDI. The same reversal occurs in the two Stream 6 comparisons with pretreatment, whereas the directions agree in the remaining eight comparisons. Neither ranking similarity nor a reversed direction establishes a more accurate ecological assessment; both describe what the chosen weighting does.

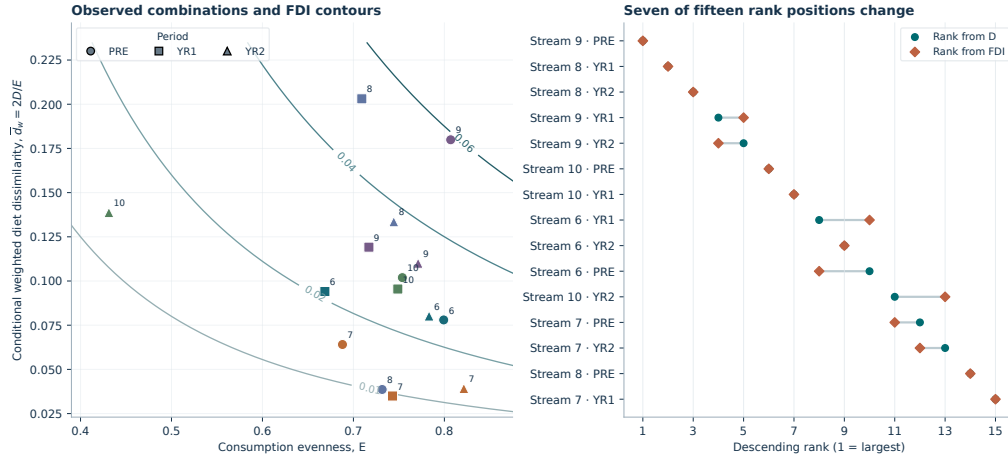


Figure 11: What the extra evenness weighting contributes in the 15 corrected stream–years. Left: observed consumption evenness against conditional consumption-weighted dietary dissimilarity $\bar{d}_w = 2D/E$; dashed contours show the exact product $FDI = E^2\bar{d}_w/2$. Right: ranks by weighted dietary component D and by full FDI for the same stream–years. Connected rank marks highlight modest changes rather than an independent metric validation.

Table 7: Corrected within-stream changes relative to PRE. FDI and matched FDI changes are absolute differences; consumption changes are percentages of pretreatment total intake.

Stream	ΔFDI_{YR1}	ΔFDI_{YR2}	ΔC_{YR1} (%)	ΔC_{YR2} (%)	$\Delta FDI_{YR1}^{matched}$	$\Delta FDI_{YR2}^{matched}$
6	-0.00388	-0.00031	+78.8	+12.8	+0.00252	+0.00686
7	-0.00552	-0.00194	+18.4	+19.0	-0.00501	-0.00128
8	+0.04073	+0.02665	+40.1	+60.8	+0.02500	+0.02625
9	-0.02794	-0.02584	+33.0	+55.5	-0.02919	-0.02562
10	-0.00219	-0.01606	+41.4	+180.9	-0.00211	-0.01654

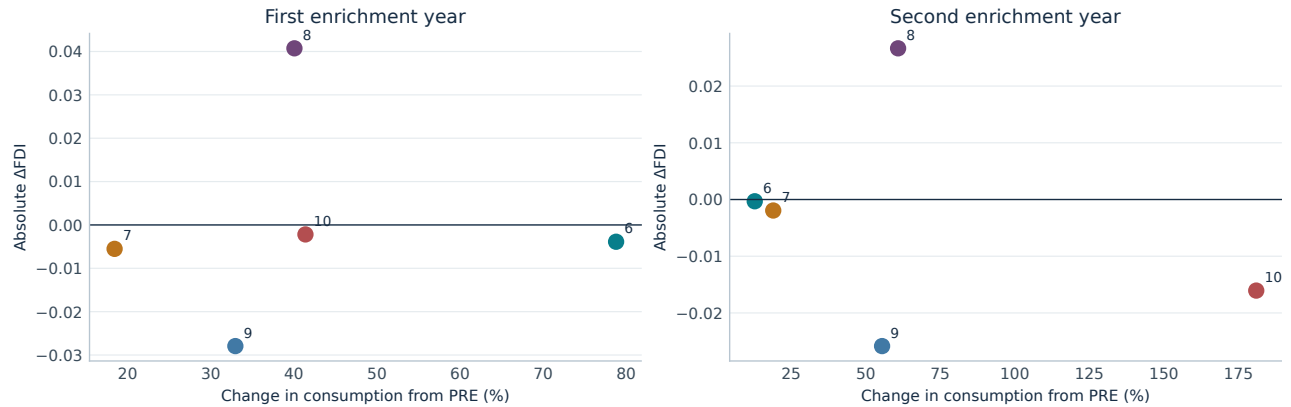


Figure 12: Within-stream divergence between change in total consumption and change in FDI. Numbers label streams. This is not an independent validation test because both axes are derived partly from the same food-intake estimates.

6.5 Paired inference and sensitivity to taxa coverage

A two-sided exact sign test assesses only whether FDI consistently increased or decreased across the five streams. With one positive and four negative changes in each year, the two-sided exact test yields $p = 0.375$ for both YR1 and YR2. These tests are underpowered and do not account for heterogeneous dosing or time-dependent influences. Without untreated contemporaneous controls, the observed within-stream differences cannot be causally attributed to nutrient enrichment alone.

Restriction to consumers present in all three periods within each stream gives 9, 10, 10, 11, and 10 shared taxa, respectively.

Recalculation on these shared subsets preserves 8 of 10 full-community before–after change directions; the two reversals both occur in Stream 6 (Table 7). Results are therefore relatively stable to consumer coverage in four streams, but the direction of change is contingent on which consumers are represented in the fifth.

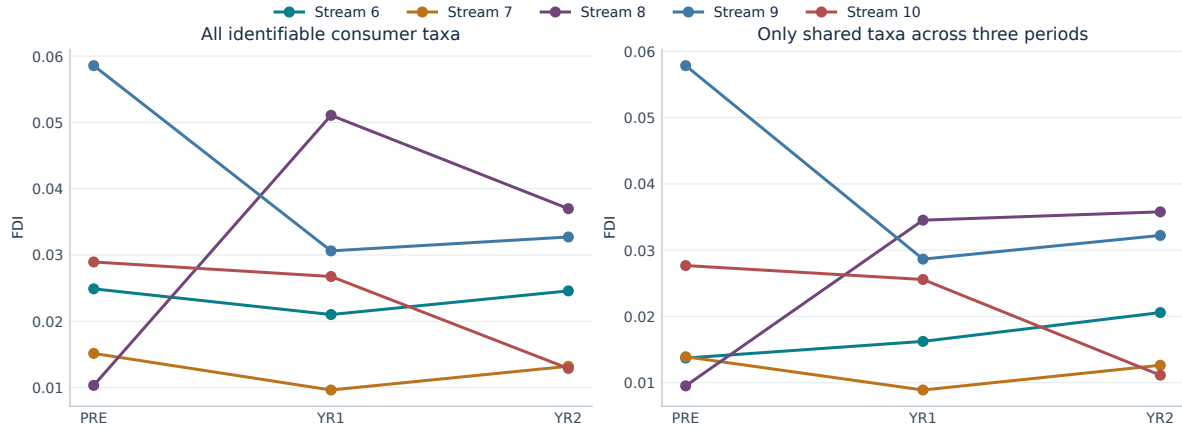


Figure 13: Recalculated full-assemblage versus shared-taxa FDI. Of ten comparisons against PRE, eight retain their direction. Stream 6 reverses in both enrichment years after restriction to taxa observed in every period.

6.6 An independent functional outcome: measured litter decomposition

Descriptive changes in an index do not establish that it predicts ecosystem functioning. A preliminary outcome comparison was therefore conducted with period-matched stream-level leaf-litter decomposition rate K obtained from the “TBoFP and Flows for Community” worksheet in the dataset archived by Demi et al. [21]. The dataset metadata attributes these rates to Rosemond et al. [26]. Fifteen observations, corresponding to five streams across three study periods, were matched to the calculated FDI values. Because these measurements come from the same experimental system, the comparison is exploratory rather than an independent multi-watershed validation.

To avoid fitting and testing on the same stream, all three observations from one stream were withheld together. For each of five folds, ordinary least squares regressions were fitted on the other four streams and tested on the held-out stream. Every model included study period (categorical YR1 and YR2 contrasts). Predictor additions included E , D , FDI, $E + D$, and $E + D + FDI$. This leave-one-stream-out (LOSO) approach tests transfer to an unseen stream within these five sampled systems, not transfer to a new ecosystem region. Results appear in Table 8 and Figure 14.

Table 8: Independent litter-decomposition outcome K : held-out-stream errors, $n = 15$ and five LOSO folds. Smaller RMSE is preferable. Neither the complete index nor joint component/product models improved the period-only baseline.

Predictors	LOSO RMSE	LOSO MAE
Study period only	0.001720	0.001347
Period + E	0.001874	0.001480
Period + D	0.001978	0.001582
Period + FDI	0.002016	0.001628
Period + $E + D$	0.002100	0.001720
Period + $E + D + FDI$	0.002614	0.001914

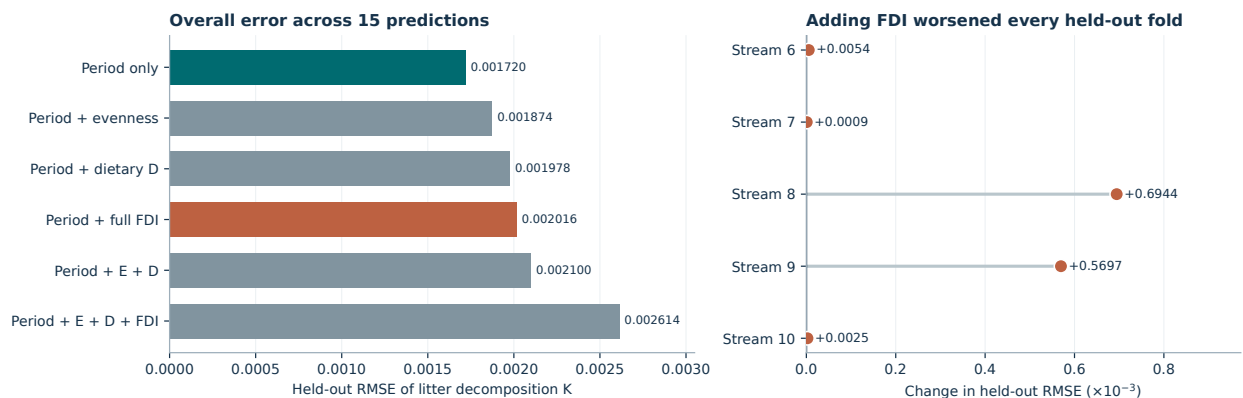


Figure 14: Held-out-stream predictive assessment for measured litter decomposition K . Left: overall leave-one-stream-out RMSE for evaluated predictor sets. Right: the change in error for each omitted stream after adding FDI to the period-only baseline; positive values mean that prediction became less accurate. Results are from 15 observations in five folds, not 15 independent validation streams.

The period-only RMSE of 0.001720 increased to 0.002016 when FDI was included (17.2% more prediction error); adding the separate components alone also increased RMSE. At the level of individual held-out streams, adding FDI increased RMSE in all five folds, although differences were very small in Streams 6, 7, and 10. This consistency warrants reporting, but five overlapping training sets do not justify a formal claim that the model will necessarily perform worse elsewhere. Over these 15 observations, the ordinary Pearson correlation between FDI and K was approximately $r = -0.014$, compared with $r \approx 0.078$ for dietary differentiation. These results fail to establish that FDI is an independently predictive measure of stream decomposition. A secondary sensitivity comparison against leaf-litter standing stock showed a small improvement for matched FDI and matched D , essentially indistinguishable (LOSO RMSE 57.199 and 57.193, respectively), so it likewise does not establish the product's incremental advantage. The small sample and repeated-measures design limit generalization.

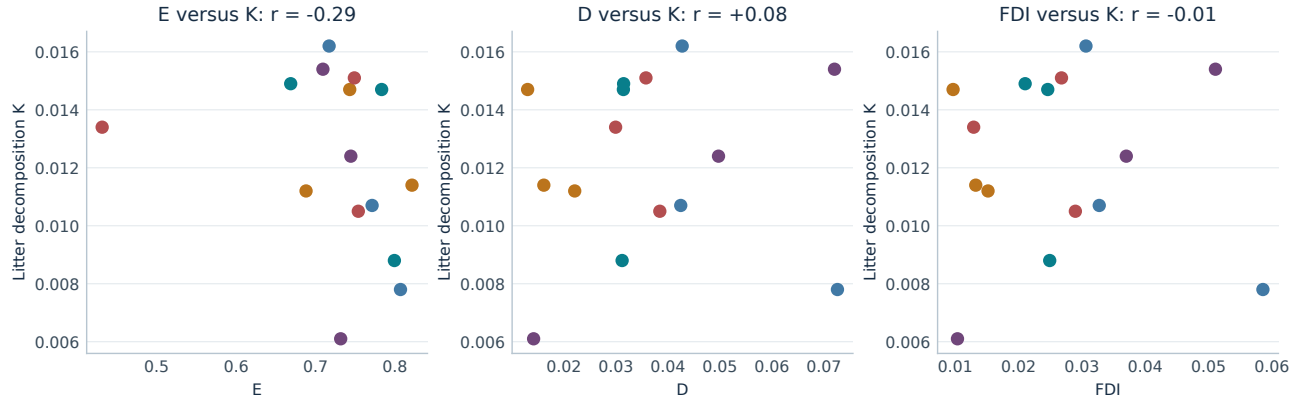


Figure 15: Descriptive relationships of K with consumption evenness, weighted dietary distance, and FDI across 15 repeated stream-years. Pearson r values are descriptive and not treated as independent replicates or causal tests.

6.7 Interpretation and measurement limits

The corrected five-stream analysis provides the most direct empirical case here for which resource-specific estimated food-intake data permit calculation of both factors of the index at taxon level. Its main descriptive lesson is that a larger amount of food moving through a stream does not tell us how that food is divided among consumers or whether their diets have become more similar. For example, an increase in total intake can occur because most taxa consume more of the same resources, because one taxon becomes dominant, or because feeding is redistributed across taxa. These scenarios can have different FDI values even when total intake changes in the same direction. Increasing total consumption did not entail increasing FDI: intake rose in all streams during the enrichment years, whereas the balance among consumers and their dietary differentiation changed in mixed directions. The distinction is mathematically expected, because multiplying every C_{iskt} within a community by the same constant leaves all normalized p_{ist} and r_{iskt} values unchanged, even though total flow increases. It remains a valid descriptive distinction, not a demonstrated independent ecological advantage.

The original experiment investigated stoichiometry, energy transfer, decomposition, and resource quality [20,21]. Its food-flow estimates share inputs with the FDI calculation, and the five treatment levels lack untreated contemporaneous controls in this reanalysis. The separately measured litter outcome allows a more stringent exploratory comparison, but the present product was not superior to the simpler predictors and period-only baseline. Subsequent work requires additional watersheds, defined biological outcome targets, diet/flow measurement uncertainty, and evaluation of the added value of FDI beyond its components and previously established Rao-style summaries.

7 Discussion

7.1 Scope of the empirical evidence

The three field datasets address related but nonidentical questions within the mathematical framework introduced in the earlier collaborative preprint [22]. The present manuscript cites that source for the NetLogo origin and original index definition but neither republishes its simulation results nor presents that work as an independent new result. The Jinsha River demonstrates how the full equation operates with modeled food-consumption fluxes and Ecopath diets; the May 2023 period has highest fish Shannon diversity and lowest FDI. More importantly, its weighting sensitivity demonstrates how a choice of consumer contribution changes FDI rankings. The Singapore study evaluates dietary dissimilarity across fish and whole-food-web scopes, but cannot calculate full FDI without consumer-level intake. The corrected experimental-stream analysis computes the complete index at taxon level in 15 stream-years; increasing consumption throughput does not fix the direction of the FDI response, and matching taxa changes two of ten before-after directional comparisons. Independent litter-outcome analysis does not support greater predictive power for the product in these data.

Table 9: Independent empirical datasets, their available inputs, and inferential boundaries. “Complete FDI” means both equation factors are computable, not that ecological prediction has been validated.

Dataset	Units and sampling	Metric computed	Main evidential constraint
Jinsha River [17]	Four periods; six Ecopath consumer groups	Complete FDI with modeled food consumption	$n = 4$ temporal summaries; biomass alternative changes highest-ranking period.
Singapore [18,19]	12 reservoirs; fish versus whole consumer webs	Unweighted pairwise dietary dissimilarity only	Consumer-level absolute intake unavailable; community scope affects rankings.
Coweeta streams [20,21]	Five streams, three periods each; 192 corrected consumer profiles	Complete FDI, shared-taxon sensitivity, independent K exploratory test	Mixed changes, 8/10 directional persistence, and no LOSO litter-outcome predictive gain.

7.2 Contributions beyond the original simulation

The original NetLogo work introduced the index and illustrated its behavior under simulated ecological conditions [22]. The present study makes a different contribution: it asks whether the required inputs can be identified in real food-web records, how much the calculated results depend on decisions made during data preparation, and whether the score is related to an ecological process that was measured separately. These questions matter because an equation can behave consistently in a simulation while being difficult to interpret when the available field data differ in taxonomic resolution, measurement units, or completeness.

The weighting comparison in the Jinsha River makes the problem concrete. The May 2023 food web ranks highest in fish Shannon diversity but lowest in FDI. Those summaries should not be treated as competing estimates of the same quantity: Shannon describes relative fish abundance, whereas FDI is calculated from modeled intake and diets of six broader consumer groups. Their disagreement establishes that the chosen trophic description is not interchangeable with a taxonomic one. The biomass alternative changes the highest-ranked period again, reminding us that the apparent conclusion depends on what is accepted as a consumer’s contribution.

The stream data distinguish a second question. Greater total intake says little about whether consumers share that intake evenly or obtain it from different resources. Within these food webs, the full index generally followed its weighted dietary component closely (Spearman $\rho = 0.971$ across 15 stream-years), but the extra evenness term changed the direction of both Stream 6 comparisons with pretreatment. The added multiplier therefore has an observable and readily explained effect, even though it does not create a wholly separate ecological signal. In Singapore, the change from fish-only to whole-food-web results illustrates a more basic restriction: an overlap summary depends on which organisms are included, and a full consumption-weighted FDI cannot be recovered when the relevant consumption totals are unavailable.

These distinctions justify reporting FDI as a compact description when the study specifically concerns the distribution of consumer activity together with differences in diets. They do not justify treating a larger score as inherently desirable. The litter-decomposition analysis is important for precisely this reason. It used an observed ecosystem process that was not used to construct the index, yet adding FDI failed to improve held-out predictions in any of the five streams. The result neither invalidates a descriptive index nor supports ecological prediction; it identifies the stronger type of evidence that further applications would have to supply.

The empirical findings also help identify when an integrated score is useful. FDI is most readily interpreted when the ecological question concerns both the distribution of food consumption among consumers and the extent to which their diets differ. Neither component alone expresses that particular combination. The stream analysis shows that the additional evenness weighting can change the direction of individual temporal comparisons, while the close association between FDI and the underlying dietary component indicates that much of their information is shared. The product is therefore a deliberate summary of two related properties, not an independent observation or an automatic improvement over reporting them separately. In applications where the components lead to different interpretations, presenting all three quantities is especially important.

7.3 Interpretation of the combined measure

The proposed FDI provides an integrated, consumption-weighted description of ecological community structure by combining the distribution of resource consumption with differentiation in dietary niche use. Its multiplicative formulation places additional emphasis on consumption evenness, distinguishing communities that differ in the balance between consumer contributions and dietary specialization. The relation $FDI_u = E_u^2 \bar{d}_{w,u} / 2$ explicitly shows why this happens: consumption weights already enter the Rao-style quadratic dietary component, so the product introduces a second evenness weighting. Prior research on Rao’s quadratic entropy and functional diversity [1,3,12,16,23–25] means that neither pairwise distance weighting nor combining multiple diversity facets should be described as wholly new mathematical ideas. The legitimate contribution here is a particular interpretable, consumption-defined synthesis and its empirical assessment under differing food-web data constraints.

Total food intake by itself cannot reconstruct consumption evenness, dietary overlap, or FDI; a common proportional increase in every consumer’s intake leaves the index unchanged. Conversely, FDI should not be interpreted as an independent measurement of those same inputs. Its strong correlation with D in the stream data shows how much of the observed pattern is shared, while the Stream 6 direction changes locate the smaller but meaningful influence of additional evenness weighting. For comparisons among communities with different numbers of consumers, aggregation rules, or dietary-resource categories, this distinction is especially important. Values are most interpretable alongside E , D , the underlying intake definition, and the set of consumers included. The added descriptive perspective is real, but its utility depends on the question being asked, not on a claim that the product is universally

preferable.

7.4 Limitations and future tests

The Jinsha four-period correlations are too few for reliable population-level inference, and diets and consumption ratios are model-derived. Singapore reservoir comparisons use heterogeneous consumer definitions and are restricted to niche dissimilarity rather than the full consumption-weighted product. The stream analysis relies on estimated gut-content/production food flows, only five repeatedly sampled streams, varied treatment levels, missing taxa, and no untreated contemporaneous streams. The index also depends on the number and resolution of retained consumers; its scale therefore cannot be compared uncritically across aggregation schemes or studies. Even the litter-decomposition comparison derives from the same experimental system; leave-one-stream-out prediction is exploratory and does not establish transfer across regions. No estimate here justifies claims that high FDI necessarily implies resilience, functional stability, environmental recovery, or healthy ecosystems.

Future work should prespecify an independent ecological outcome and sampling unit; compare FDI against E , D , $\bar{d}_w$, conventional taxonomic measures, Rao-style quadratic entropy, and total flux; quantify diet and flow estimation error; test multiple community aggregation schemes; and evaluate generalization under geographically separated or prospective validation. A null incremental predictive result should be reported even when the index provides a transparent descriptive decomposition of trophic organization. An additional limitation is that the empirical comparisons were exploratory rather than prespecified tests of a single hypothesis. The reservoir group comparisons and four-period correlations should therefore be viewed as patterns worth testing in larger, independently sampled datasets, not as definitive evidence of environmental effects. Reporting component values and analysis choices alongside the composite score will also help future researchers determine whether an observed FDI change reflects consumption evenness, dietary differentiation, or both.

7.5 Conclusion

This study extends the FDI framework introduced by Ayyanger and Vijay (2026) [22] from simulation to three independently published aquatic datasets. Complete index values could be calculated where consumer consumption and diet information were available; the Singapore reservoir records supported analysis of dietary differentiation but not the full consumption-weighted product. Across the empirical applications, the index offered a clear way to examine the balance between consumer contributions and differences in resource use. It also proved sensitive to how those contributions were weighted and which consumers were included. These findings establish a specific, limited contribution. FDI can complement taxonomic diversity and total resource-flow measures by summarizing a different aspect of trophic organization, with deliberate additional emphasis on consumption evenness. It should be interpreted alongside its components rather than as a replacement for established measures. The independent litter-decomposition analysis did not demonstrate improved prediction, and the present results do not justify interpreting FDI as a direct indicator of ecosystem health, resilience, or stability. The next step is to test whether the descriptive differences identified here have consistent ecological consequences in larger datasets with independently measured outcomes.

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