**Trophic interactions modify the temperature dependence of community biomass and ecosystem function**

**Running head: ecological effects of temperature**

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Research paper**Abstract**

Aquatic ecosystems worldwide continue to experience unprecedented warming and ecological change. The metabolic theory of ecology predicts that warming increases net ecosystem oxygen fluxes by increasing per capita metabolic rates. In addition, temperature change can alter species interactions, which in turn influence ecosystem function. Surprisingly, the role of species interactions in modifying the general metabolic scaling effects of temperature at the ecosystem scale remains unclear. We experimentally tested the hypothesis that cascading trophic interactions modify the temperature-dependence of community structure and net ecosystem oxygen fluxes over a range of temperatures from 19-30 °C. In a series of independent freshwater ecosystems we found that at higher temperatures, cascading effects of predators on zooplankton prey and algae were stronger. Standing phytoplankton biomass declined by 85-95% (< 1-fold) over the temperature gradient when grazing was weak or absent, and by 3-fold when grazers were present and lacked predators. These temperature-dependent species interactions and consequent community shifts only modestly affected the temperature dependence of net ecosystem oxygen fluxes. The exponential increase in net ecosystem oxygen flux over the temperature gradient, as well as monotonic declines in phytoplankton standing stock, suggested no threshold effects of warming across systems. We conclude that species interactions can modify effects of temperature on primary producer biomass and net ecosystem oxygen consumption relative to first-order metabolic theory predictions, but even so, temperature can have continuous, positive effects on ecosystem fluxes, consistent with patterns based on large-scale, macroecological comparisons. Changes in community structure, including temperature dependent trophic cascades, may be compatible with prevailing and predictable effects of temperature on ecosystem functions related to fundamental effects of temperature on metabolism.

**Introduction**

Temperature affects metabolic rates of all organisms, thereby affecting ecological patterns and processes across scales of organization – from individuals to ecosystems. Increasing temperature accelerates major metabolic processes that drive net ecosystem production and ecosystem respiration in aquatic and terrestrial ecosystems [1-4]. Highly conserved metabolic processes - photosynthesis and aerobic respiration [5] - power somatic growth, maintenance and activity in aerobic organisms. As a result, the effects of temperature on photosynthesis and respiration have accurately described the exponential increases in ecosystem-scale NPP and ER in aquatic systems across macroecological thermal gradients, after accounting for body size, nutrient content and light availability [4,6,7], suggesting that the temperature dependence of these fundamental metabolic processes constrains higher order ecological responses to temperature. The ecological importance of temperature-dependent *per capita* metabolic rates has led to the use of metabolic models, and the development of an important theme in metabolic scaling theory (MST) of general temperature dependence, to understand and predict ecological change across scales, from local to global [3,4,8]. Models that associate change in ecosystem scale metabolism (e.g., oxygen or carbon flux) with individual-level oxygen production and respiration, but bypass the complexity of population and community dynamics, provide much needed predictability for how climate change affects ecosystem functions when ecosystems are compared across broad spatial or temporal thermal gradients [2,4,9].

One challenge has been to reconcile the high explanatory power of general temperature dependent metabolic scaling models at macroecological scales with the well-documented contingencies of how temperature affects community level outcomes of population dynamics and species interactions [10-16]. Whether at macro-ecological or community (e.g., single-site) scales, ecosystem-level functions (ER, NEP) or biomass is simply the sum of per capita function (respiration, net photosynthesis) and biomass. Yet, at local scales, it is well known that species interactions can strongly influence community-level NEP, ER and primary producer abundance, such that simply scaling up a sample of observed per capita responses to temperature does not adequately predict community level responses. Experiments have shown that temperature increases can positively or negatively affect productivity, and other properties associated with ecosystem function such as biomass, abundance and species composition (O’Gorman, Cross, Padfield, etc). Population dynamics and species interactions can themselves change ecosystem functions [17]. This paradox between macroecological patterns – which can be consistent with direct scaling of per capita thermal responses - and experimental results that are more complex leads to the suggestion that general metabolic scaling models that do not consider the complexities associated with species interactions do not apply at the local scales. Reconciling these apparently divergent patterns is critical to improving understanding and projections of how shifting global thermal regimes affect ecological patterns and processes across scales and achieving a more unified understanding of ecology across scales.

One way to reconcile the apparent context dependence of empirical results under controlled conditions with the generality of temperature dependence of ecosystem function at broader scales is to consider how the direct and indirect effects of temperature on demographic processes interact to produce patterns that appear to contradict the general relationships between temperature and ecosystem functions at macroecological scales [11,13,18-21]. Direct effects of temperature on *per capita* metabolic rates cause organismal photosynthesis and respiration rates to increase exponentially as temperatures increase when resources are not limiting. This relationship between temperature and fundamental metabolic rates (photosynthesis and respiration) is referred to as general metabolic scaling [1]. For any single phenotype, above some optimal temperature signals of metabolic scaling become overwhelmed by stress responses and declines in performance and metabolic scaling no longer explains the effects of temperature. However, in multi-species communities the signal of metabolic scaling is likely to dominate over a broad range of temperatures if species’ with distinct thermal phenotypes can compensate for each other along the thermal gradient [18,22]. Warming is also associated with other biological changes that affect species interactions, such as reductions in body size (the temperature size rule, [23,24]), fecundity, and attack rates (Fig 1) {Anonymous:2011hx, Anonymous:2018bn, Anonymous:2011tb, Sentis:2015dt}.

The temperature-dependence of consumer resource interactions has received substantial attention in this context, because these *trophic* interactions can influence many aspects of community structure and ecosystem function, including biomass, abundance, species composition and stability [10,28]. Trophic species interactions appear to strengthen with warming. Series of trophic interactions, called *trophic cascades* (Fig 1), link predator populations to the abundance, biomass and ecosystem functions of primary producers [17,29] and the strength of trophic cascades depends on body size, primary production (Delong et al 2015). Considering the multitude of indirect effects of temperature on population dynamics and the prevalence of consumer-resource interactions and trophic cascades in aquatic systems [30] begs the question, how is it that these locally dominant population-level responses to temperature [31,32] do not appear to cause major variation or context dependence in macroecological relationships between subcellular metabolic processes (photosynthesis, respiration) and ecosystem processes (NEP, ER)?

Here, we aimed to resolve the paradox between apparent direct effects of temperature on ecosystem functions (NEP, ER) that emerge when comparing communities across larger gradients and the potentially more complex effects of temperature at the population and community scales over time. We experimentally tested the hypothesis that temperature-dependent trophic interactions in a trophic cascade alter the effect of temperature on community properties such as biomass, abundance and body size, but have little or no effect on the temperature dependence of ecosystem functions (NEP, ER) over a temperature gradient (Fig 1). In freshwater plankton communities, we compared the effects of temperature on community properties typically measured in warming experiments with the effects of temperature typically compared in macroecological studies under controlled conditions to minimize all variation among ecosystems except that due to trophic structure (presence of consumers and predators) and temperature (Fig 1). We quantified ecosystem function (NEP, ER) and community structure (biomass, abundance) across an experimental temperature gradient of 10 °C. We found that exponential effects of temperature on algal biomass were much greater than effects of temperature on NEP and ER, suggesting that even large changes in community structure do not necessarily lead to large changes in how temperature affects NEP and ER.

**Hypotheses**

**Metabolic theory framework:**

We drew on the Metabolic Theory of Ecology (MTE) frame our hypotheses and predictions for how temperature affects NEP and ER via per capita metabolic temperature dependence and indirect effects of temperature at the community scale. We first briefly outline the framework, and then express our specific hypotheses. MTE relates whole-organism metabolic rates (*bi*) and related biological functions for organism *i* to body size (*mi*) and body temperature (*T*, in Kelvin) [1,33,34]:

Eqn 1

in which activation energy (*E*a, in eV) captures the exponential effect of temperature on per capita metabolic rate*, k* is the Boltzmann constant (eV/K), *b0* is a normalization constant independent of body size and temperature that includes the effects of temperature-independent traits on metabolic rate. The allometric scaling factor α relates metabolic rate to body size [see methods for justification of Eqn 1 over alternative models].

The effects of temperature on ecosystem metabolic rates (*BR*), such as NEP or ER (mgO2/L/hr) reflect the sum of all *per capita* photosynthesis rates by autotrophs and respiration rates by autotrophs and heterotrophs as well as shifts in abundance, body size and acclimation. Following Barneche et al (2014), we capture these direct and indirect effects of temperature on ecosystem scale metabolic rates in the following equation (see Barneche et al 2014 for derivation):

. Eqn 2

The term captures the temperature dependence *ER* (eV) of ecosystem-level metabolic rate *R*. Equation 2 represents a ‘first order metabolic scaling’ prediction that ecosystem scale mass-normalized metabolic rates (e.g., NEP) vary proportionally with the temperature dependence of the underlying metabolic processes. Observed temperatures *T* are related to an arbitrarily chosen reference temperature *Tc*. These studies assume that *b0* and *MB* to not vary with temperature in influential ways [1,18].

When considering the indirect effects of temperature on ecosystem oxygen production and respiration, we can consider how the other terms in Eqn 2 may vary with temperature. To account for changes in total biomass, body size or relative abundance of differently sized individuals associated with temperature, we use the term . The total biomass, *MB*, equals the sum of mass *mi* for all individuals *i* to *J* in volume V . The term is the average of all individual metabolic biomasses, , corrected for the greater contribution to total mass-specific metabolic biomass by small individuals resulting from the allometric scaling () of oxygen production and consumption with body size [2,18]. This ‘mass correction’ is necessary, because if community biomass is comprised of one large individual, that biomass will [produce and] consume less oxygen per gram biomass in a given time period than if the same total biomass were comprised of many small individuals; in other words, increases *mii* declines and density increases. Finally, the term relates the normalizing constant *b0* to a particular temperature (here, *Tc*), allowing this to change with temperature. If thermal traits acclimate or species composition shifts with temperature, this term would capture that change.

Hypotheses

1. ***Trophic interactions modify the effect of temperature dependent metabolic rates on total algal biomass.*** Total algal biomass can be expressed in terms of temperature, traits and size distributions by rearranging eqn 2:

Eqn 3

If we assume that are independent of temperature, we predict that algal biomass *MB* declines with temperature by *ER*, in this case *ER* = *ENEP*. This prediction has been supported empirically in single species algae system [21], and in that system the predicted decline in total abundance was robust to changes in cell size. However, it is unlikely that grazers would not alter the abundance and size of algae, altering among trophic treatments, and also the traits of algae, and thereby modifying among trophic treatments. A fuller integration of how temperature and trophic treatment affect these terms would require new theoretical development and is beyond the scope of this paper, but we use the equation here to highlight why we expect trophic structure and temperature to affect algal biomass. To test this hypothesis, we compare *MB* trends with temperature across ecosystems with and without a trophic cascade.

1. ***Increasing temperature strengthens the trophic cascade***. We estimated the strength of the trophic cascade as the ratio of primary producer biomass in the presence of predators (AGP) *vs* in predator-free environments (AG) (Shurin et al 2002). We predicted that predators would reduce the abundance of zooplankton through predation, and shift zooplankton composition to smaller sizes and less-edible species, typical of classic freshwater trophic cascades [35], and that these trophic interactions would strengthen with higher temperatures due to the effect of temperature on *per capita* grazing rates. We can relate algal biomass among treatments using Eqn 3 for primary producer biomass in the presence of predators (AGP) and grazers only (AGP), simplifying and taking the log to yield (see Methods, eqns 7-10, for details):

Eqn 4

The strength of the trophic cascade is therefore expected to increase with temperature reflecting the relationship between the temperature dependence of acclimation, compensation and body size shifts. But because trophic interactions could affect the realized temperature dependence via several possible mechanisms – shifts in body size, traits, etc, - it is not possible to predict *a priori* how to model this change for this system.

1. ***Temperature dependence of net ecosystem production and respiration depends on the strength of the trophic cascade.*** We test this by using Equation 2 to model our results, but we allow *bo(TC)* to vary not only with temperature but with trophic structure (*Z*), and we expect that trophic structure will influence the number and size of individuals, and thereby affect . Alternatively, trophic structure may not modify the relationship between temperature and R, as observed in broad-scale macroecological comparisons. We can test these alternate predictions by comparing models with and without *bo* and terms that depend on ecosystem temperature and trophic structure.

To summarize, for each hypothesis, we used linear mixed effects models (LMMs) to test ‘first order’ metabolic scaling models for the appropriate model (equation 2, 3 or 4) that included *bo* as independent of the ecosystem’s temperature or trophic structure [Methods]. We tested alternate models that allowed *bo* to vary with ecosystem mean temperature, weekly temperature and/or trophic structure. If the simpler, first order models are best supported, we would infer that indirect effects of temperature do not overwhelm the signals of direct metabolic scaling effects on ecosystem functions, consistent with inferences drawn in macroecological studies. Our data do not permit testing predictions about size distributions or trait shifts, but support for models variation in *bo*and among treatments would suggest these mechanisms as likely explanations.

Results

As temperature increased across ecosystems, phytoplankton biomass, estimated as the concentration of *chlorophyll a* in the water column, declined (Fig 2A). Trophic interactions modified the effect of temperature on *chlorophyll a* concentration (Fig 2A, Table 1). This inference is supported by the inclusion of a main effect for temperature in the best model (Table 1) and an estimate for the temperature dependence (Fig 3). Phytoplankton biomass declined much more strongly with temperature in algae-grazer (predator-free) communities, with a decline of over three orders of magnitude in phytoplankton biomass standing stock over the 10 °C temperature gradient (Fig 2A). In the algae-grazer-predator treatments, phytoplankton biomass declined with a slope indistinguishable from that in the algae only treatments (Fig 3). We observed shifts in taxonomic composition with temperature, suggesting support for temperature dependent *b0*, and we cannot reject temperature dependent shifts in size distributions within or among species with the data we have. [I may be able to add more results on PP taxonomic shifts and size]

### **Table 1. Model selection results for phytoplankton biomass.** We used linear mixed effects models with terms for ecosystem trophic structure (*Zj*), weekly average temperature (Tij), and average temperature over the entire experimental period (TM), and with ecosystem identity as a random effect. We compared models using likelihood ratios (LogLik), AICc, Akaike weights (w) and delta AICc weights.



Consistent with our second hypothesis, and the patterns observed for phytoplankton biomass, there was a strong trophic cascade in the warm ecosystems by the end of the experiment (Fig 2B). The trophic cascade became apparent after the first weeks of the experiment, and strengthened over time and with temperature (Fig 2B, Fig S2.7) (Table 2). The best model included a term for mean ecosystem temperature (*TM*), as well as week, and a week x temperature interaction. By week 9, the log response ratio of chlorophyll *a* concentration with vs without predators increased exponentially with temperature (Fig 2B) to an estimated *ETC* = 0.77.

### **Table 2. Model selection results for trophic cascade analysis**. We used linear mixed effects models with terms for average temperature over the entire experimental period (TM), week 2-9 (Wk) and their interaction. We treated the power level (e.g., 100W, 200W, etc), our temperature treatment, as a random effect to account for repeated measures on ecosystems over time. We compared models using likelihood ratios (LogLik), AICc, Akaike weights (w) and delta AICc weights.



We find additional evidence of temperature dependent trophic interactions in the responses of the zooplankton grazer assemblages to warming and predation. Predators and increased temperature independently reduced total zooplankton density (Table 3); zooplankton density declined with increasing temperature (*EZP* = 1.28 95% CI: 0.19 – 2.39, z = 2.31, p = 0.021 based on regression with negative binomial distribution) (Fig 4). Predators reduced densityof *Daphnia*, the dominant grazer (linear regression with Poisson distributed errors: estimate: = -1.14 95% CI: -1.91, -0.36, z = -2.87, *P* < 0.01) and density declined with increasing temperature (Table 4) although the temperature term retained in the best model was not significant based on the p-value (*ED* = 0.88 95% CI: -0.59, 2.35, z = 1.17, *P* = 0.24). Copepod density declined with temperature (*EC* = 2.21, z = 3.12, *P* = 0.002), and not in response to predation (best model did not include a predation term, Table 6). We observed no significant effect of temperature or predation on the total zooplankton community body size distribution (model estimated mean *Daphnia* body size was 0.83 cm 95% CI: 0.79-0.88, and mean copepod size was 0.54 cm + 95% CI: 0.52-0.57; Fig S3.8).

**Table 3: Zooplankton density**. Results of model selection for zooplankton abundance in ecosystems with grazers and with grazers and predators. We used negative binomial regressions with ecosystem identity as a random effect [Methods]. Models included terms for weekly average temperature (Tij), ecosystem trophic treatment (*Zj*) and their interaction, and a random effect for ecosystem identity. We compared models using likelihood ratios (LogLik), AICc, Akaike weights (w) and delta AICc weights.



**Table 4:** Daphnia density: Results of model selection for *Daphnia* abundance in ecosystems with grazers and with grazers and predators. We used Poisson regressions with ecosystem identity as a random effect [Methods]. Models included terms for weekly average temperature (Tij), ecosystem trophic treatment (*Zj*) and their interaction, and a random effect for ecosystem identity. We compared models using likelihood ratios (LogLik), AICc, Akaike weights (w) and delta AICc weights.



**Table 5:** Copepod density: Results of model selection for copepod *spp* abundance in ecosystems with grazers and with grazers and predators. We used Poisson regressions with ecosystem identity as a random effect [Methods]. Models included terms for weekly average temperature (Tij), ecosystem trophic treatment (*Zj*) and their interaction, and a random effect for ecosystem identity. We compared models using likelihood ratios (LogLik), AICc, Akaike weights (w) and delta AICc weights.



Across ecosystems, and in the context of temperature-dependent trophic cascades and community taxonomic shifts, higher temperatures increased net ecosystem oxygen production (NEP) and respiration (ER) (Tables 5, 6, Fig 5). The lmm for NEP (Table 5) and ER (Table 6) suggested that ecosystem temperature and trophic structure interact to influence ecosystem oxygen fluxes, yet their estimated temperature dependences did not appear to differ when confidence intervals were compared (Fig 3). The estimated across-system temperature dependence of NEP was the strongest in algae-only communities (Fig 5), and confidence intervals for the temperature dependence term include 0 for the systems with predators (Fig 3). Ecosystem respiration (ER) increased with temperature across ecosystems (Fig 5) and only the estimated temperature dependence in systems with predators was weak or absent (Fig 3).

Table 5: Results of model comparisons for effects of temperature and time on net ecosystem productivity (NEP) based on AIC weight (*w*) and δAICC values. Nested versions of the full model (Eqn 11, Methods). Response variables are modelled as functions of temperature *Tij* for each tank *j* on week *i* relative to the mean temperature for tank *j* over all weeks (T in Kelvin), and food chain length (*Zj*). Models included a random effect for the experimental unit – tanks with and without predators receiving the same power inputs. See Methods for additional details on modeling.



Table 6: Results of model selection for effects of temperature and time on ecosystem respiration (ER) based on AIC weight (w) and δAICC values. Nested versions of the full model (Eqn 11, Methods). Response variables are modelled as functions of temperature Tij for each tank *j* on week *i* relative to the mean temperature for tank *j* over all weeks (T in Kelvin), and food chain length (*Zj*). Models included a random effect for the experimental unit – tanks with and without predators receiving the same power inputs. See Methods for additional details on modeling.



In addition to the variation among ecosystems in temperature that was the main focus of our analysis (Methods), temperature varied within our experimental ecosystems over time (Fig S1). Effects of temporal temperature variation on phytoplankton biomass differed starkly from effects of temperature among ecosystems (Fig 6). Within ecosystems, higher temperatures were associated with higher phytoplankton standing stocks, contrary to the among-ecosystem patterns of lower biomass at warmer temperatures. Within ecosystems, effects of temporal temperature variation depended on trophic structure, with the strongest effects of within-ecosystem temperature variation apparent in the AG treatments (Fig 6, top row). Net ecosystem oxygen production (NEP) varied with temperature within ecosystems, and there is some evidence that this temperature effect interacted with both the trophic structure treatment and the overall average ecosystem temperature (Table 5). Net ecosystem respiration varied within ecosystems over time, but this variation did not depend on temperature treatment or trophic level (Table 6).

**Discussion**

The joint effects of biodiversity loss and climate change are affecting ecosystems worldwide [36,37]. Biodiversity loss can change species interactions in a community [30], and the role of local species interactions in amplifying or moderating effects of temperature on ecosystem functions is a critical link between biodiversity at the community level and climate change impacts on individual and ecosystem level performance [38,39]. Here, we found that in a series of aquatic ecosystems characterized by the presence or absence of predator-prey species interactions, temperature-dependent trophic cascades only modestly altered the effects of temperature on net ecosystem oxygen production and consumption (NEP and ER). These two ecosystem functions (NEP, ER) reflect the total metabolic activity of all aerobic organisms in the ecosystem. They are directly related to carbon storage and cycling by aquatic ecosystems [40], and they are expected to vary with temperature reflecting the effects of temperature on per capita metabolic rates (Eqn 1). We found that higher average temperatures increased net ecosystem oxygen production and consumption while total phytoplankton biomass declined. Local community structure and species interactions associated with a typical food web motif in freshwater food webs – the trophic cascade - modified how temperature affected oxygen flux, thereby linking changes in biodiversity ecological responses to climate warming. Predators reduced the grazing pressure imposed by the dominant grazer, *Daphnia*, by reducing its density, and thereby shifted grazer assemblages toward the less effective copepod grazers. This trophic cascade, mediated by shifts in grazer composition as well as total density, is a classic food web motif in freshwater systems [35].

These results also provide a key test of how we can use one of the more general ecological theories for temperature effects – the metabolic theory of ecology – to understand community-level responses to temperature change. The effects of temperature on ecosystem function varied with the types of species interactions dominant in each local community. The temperature dependences of net ecosystem fluxes were least pronounced in communities with grazers and predators. These results led us to reject the ‘first order metabolic theory’ hypothesis that temperature dependence of ecosystem functions scales directly with general temperature dependence of metabolism and further suggests that changes in species interactions within communities, such as loss or gain of a predator species, could alter the responses of net ecosystem fluxes to temperature changes.

The metabolic theory of ecology predicts that highly conserved metabolic rates (respiration, photosynthesis) are sensitive to temperature in ways that emerge at scales of communities and ecosystems. Across broad spatial scales, and within experiments, ecosystem level fluxes have been shown to vary with temperature according to the temperature dependences of photosynthesis and respiration, irrespective of trophic structure and consistent with the ‘first order metabolic scaling’ hypothesis (Fig 1ii) [2-4]. Despite repeated support at macro-ecological scales, this ‘first-order metabolic scaling’ prediction has been rejected or challenged at local community and ecosystem scales [41,42]. Other studies have found that acclimation and adaptation can compensate for temperature effects on fundamental metabolic processes at the ecosystem scale [9,22,43], and additional theory has been developed to explain how the general temperature dependence of metabolic rate interacts with population and community level processes to influence the emergent responses to temperature, producing an expanded version of metabolic scaling theory that incorporates local ecological and evolutionary processes [11,13,18,44,45]. Here, we contribute to this line of inquiry with our finding that the way net oxygen fluxes varied over the temperature gradient in this controlled experiment depended on trophic structure of the community. Our models indicated the need for the species interaction x temperature model term. Still, we observed temperature dependences that were consistent with expected temperature dependence of the underlying metabolic processes of photosynthesis (*Ea* = 0.32 eV) and respiration (*Ea* = -0.65 eV), though our confidence intervals on estimates are wide enough to preclude strong inference about the exact values of the activation energies in this study.

We observed a much stronger decline in phytoplankton biomass with warming than the increase in oxygen fluxes, as indicated by higher slope values for biomass (Fig 3). This difference in phytoplankton biomass and oxygen-flux responses to temperature could reflect several processes operating at different scales of organization. First, we expect that per capita rates of oxygen flux increase with warming, so that a given biomass of phytoplankton can be more productive at warmer temperatures if resources are not limiting [4,11,46]. In addition, if size distributions shift toward smaller cells, as is common with warming [47,48], the allometric scaling of metabolic rate with body size (Eqn 1b) predicts greater oxygen flux for a given biomass. At the ecosystem scale, resource supply may change with temperature, if microbial assemblages, biological nitrogen fixation, and recycling processes accelerate [49], creating a resource gradient in parallel with the temperature gradient [10,46]. Additionally, benthic algae may contribute to NEP and ER estimates [50]. Though we did not observe notable amounts of accumulated benthic algae in our tanks, even small amounts could have contributed to total ecosystem fluxes and led to covariation in total biomass with temperature. If the ratio of phytoplankton to benthic algae was temperature-dependent [50], our primary producer biomass estimates may have increasingly under-represented total algal biomass at higher temperatures. To be conservative, we did not present mass-normalized NEP estimates because we could not normalize to any benthic algal metabolic biomass. Covariation between biomass and temperature is common across geographic variation in temperature [22,43,49] and therefore present in other estimates of NEP across broad spatial scales when biomass cannot be estimated well. We cannot distinguish among these explanations in our experiment, and we suspect they are all relevant.

We observed no sign of ecosystem collapse or threshold responses to warming. Changes in community structure and the increase in trophic control along the temperature gradient appear to be exponential and monotonic over the 10 °C gradient (Eqn 1b, Figure 3), suggesting that linear (or additive) models of temperature effects in most warming experiments, which only test two or three temperatures, may underestimate warming effects over broader thermal gradients (Fig 3D, Fig S3.3, S3.4). We observed little evidence of abrupt transitions that might be expected if thermal stress responses by individual phenotypes drove ecosystem scale responses. While individuals may experience thermal stress and decline in performance at high temperatures, in our systems these effects were functionally compensated for by other species and increases in per capita performance. The exponential effects of temperature on biomass and oxygen fluxes persisted for all species interaction scenarios. In other studies, in the absence of grazers, algal biomass tends to decline with increasing temperature when resources do not increase with temperature [11,51,52]. This is an expected consequence of increased mass-specific metabolic demand at higher temperatures, and could be exacerbated by temperature-dependent consumer control of phytoplankton biomass [10,11]. Though we did not observe signs of transient dynamics in these communities over time, we also cannot conclude that these systems had reached an equilibrium or stable state. Longer experiments have demonstrated continued shifts in community composition after months and years of warming [53,54].

In our systems, community biomass and abundance in food webs were more resistant to community change with warming and longer food chains. Predators reduced zooplankton density and caused a clear trophic cascade. Trophic control, and therefore any mitigating effects of predators on biomass change, was weak at low temperatures and increasingly strong at higher temperatures (A vs AG treatment, Fig 3D). This pattern is consistent with previous findings that ecosystem functions in systems with two (or even numbers) of trophic levels tend to be more sensitive to warming than systems with odd numbers, due to cascading effects of predation on primary producers [51]. Yet, this result contradicts theories in which dynamically responsive predators can make three-trophic-level systems dynamically less stable than shorter food chains [55]. The difference between the prediction for instability in population dynamics and stability in ecosystem function may be explained by biodiversity in our systems and functional compensation among zooplankton or phytoplankton species [56]. Additionally, in our experiment, predators were not dynamically responsive; they did not have time to reproduce during the experiment. Consequently, they represent mortality for zooplankton that may have varied with temperature effects on per capita predation rates by predators, but not demographic response. In many systems, predators are subsidized by other habitats and food sources, and their populations are not dynamically coupled to prey. In fact, this decoupling has been shown to be important in thermally stratified lakes [57]. Inferences drawn based on this experiment about how species interactions affect community and ecosystem responses are restricted to systems with dynamics in the primary producers and primary consumers, with fixed predation-related mortality imposed by a third trophic level.

The growing literature of experimental tests of how warming affects interacting species aims to reduce uncertainty in projected changes associated with climate change. Warming experiments have shown a wide variety of effects on species interactions, from shifts in community composition, strengthening top-down control, and shifts in body size [50,58,59]. We have shown that these shifts do alter the effects on the temperature dependence of net ecosystem oxygen fluxes. By measuring these responses over a broad thermal gradient, we have provided empirical evidence for nonlinear effects of temperature at the community and ecosystem level. Further, we have measured these changes in communities in which changes in species composition and community structure occurred within the context set by temperature constraints on energy fluxes via fundamental metabolic processes. To extend our findings to a conjecture about implications for climate change, we suggest that conservation actions that maintain predators and top down control may also promote an ecosystem that changes less with temperature than a system with a large abundance of grazers. Taken together, these results suggest our efforts to predict community change with warming may benefit from the general metabolic scaling theory framework to understand even local-scale effects of temperature change at the community level.

**[add this:**

It is well established that temperature dependence of aerobic respiration is approximately *ER*= ~0.65 eV, and that this value emerges at the ecosystem scale, such that ecosystem respiration varies with temperature across systems as *EER* = ~0.65 eV (Yvon-Durocher et al 2012, Allen et al 2005, L-U 2006, L-U 2008). The temperature dependence of photosynthesis at suboptimal temperatures appears to be *EPS*= ~0.32 eV, and this can emerge at population () and ecosystem scales () in aquatic systems, suggesting *ENEP*= ~0.32 eV, but other studies have found evidence for stronger or weaker values of *ENEP,* ranging from 0 to 1.2 eV at population and ecosystem scales (michaletz, yvon durochers, Barneche et al 2014).]

Our experimental communities shared environmental conditions, species pool and weather over the course of the experiment, differing only in temperature and trophic structure. Comparisons across ecosystems in this experiment could be considered as controlled comparison across sites, analogous to a macroecological comparison in which all other conditions are known to be similar except the one of interest (temperature, trophic structure).

Our a priori hypothesis was that EM = ENEP = 0.32 eV, which is outside the confidence intervals of our observed temperature dependence estimates. This difference could be explained by temperature dependence in the mass-normalized biomass term, reflecting temperature dependent shifts in phytoplankton size distribution or phenotype (either associated with acclimation or taxonomic shifts). While we do not have data to directly distinguish among these explanations, we do have observations of phytoplankton taxonomic composition (from subsamples), and from that can crudely estimate shifts in size based on a literature search of typical sizes. Because the trophic cascade strength could reflect both direct and indirect effects of temperature and multiple temperature dependent processes, we cannot explain this value of ETC. *We will discuss this more in the discussion.*

Although the lmm analysis did not clearly reject the hypothesis that trophic structure does not modify ecosystem level temperature dependence, the temperature dependence terms (ENEP, EER) varied much less among trophic treatments than did EM (0.99 – 1.3 for NEP, 1.3 to 0.3 for ER, and 1.3 vs 3.14 for Mb).

**Methods**

**Experimental Food Webs**

We assembled freshwater food webs in 30 outdoor mesocosms (370 L tanks) at the University of British Columbia, Vancouver, Canada (49°14’52” N, 132°13’57” W). Mesocosms were filled with municipal water on June 26th, 2012, heaters were added, and filled tanks were left for one week to allow chlorine to evaporate before organisms were introduced. We experimentally manipulated temperature (10 levels) and species interactions associated with different food chain lengths (3 levels: algae-only, algae + grazer, and algae + grazer + predator food chains, Fig. S1A-B). Tanks were arranged randomly in space with regard to treatment. The spatially randomized assignment of temperature and trophic treatments eliminated systematic variation in negligible allochthonous carbon inputs.

After 1 week, mesocosms were inoculated with pondwater (1L) from the UBC Pond Facility, containing living algae, collected and filtered through a 64-µm sieve to remove zooplankton and larvae. Three days later, we collected zooplankton at Trout Lake, Vancouver, B.C. (49°15’23” N, 123°03’44” W), with a vertical tow net (64-µm mesh). Zooplankton were mixed in buckets to homogenize species composition, were gradually introduced to mesocosm temperatures over a 12-hour gradual acclimation period to avoid stress associated with an abrupt temperature change, and dead organisms were removed. Initial experimental communities consisted of 25 phytoplankton taxa (Table S1.2), and those with zooplankton included predominantly 2 zooplankton taxa (the cladoceran *Daphnia* s*p.,* and calanoid copepod *Eurytemora* sp*.*) and, rarely, cyclopoid copepods. To ensure colonization of grazing zooplankton, in addition to the random aliquot of zooplankton added to each zooplankton ecosystem (all algae-grazer and algae-grazer-predator ecosystems), we added two individuals of *Daphnia* sp. and ten *Eurytemora* sp. Thus each zooplankton community began with *at least* 12 grazing zooplankton individuals. We introduced 2 individual notonectid predators (*Notonecta undulata*), collected from ponds at the UBC Pond Facility, on July 4th, 2012 (experiment day 8) to 10 algae-grazer-predator tanks. Notonectids generate trophic cascades by suppressing zooplankton [60]. Notonectids did not reproduce during the experiment, and we replaced dead notonectids during the experiment with similar-sized individuals from the same source population.

**Temperature and Nutrient conditions**

We added 160-µg NaNO3 L-1 and 10-µg KH2PO4 L-1 to each tank (16:1 N:P) on July 3rd, 2012. These quantities of nutrients represent typical deposition inputs to similar lakes [61]. Water was heated with submersible aquarium heaters (50, 100, 150, 200, 250, 300, 350, 400, 450 Watt) to increase temperature above ambient daily temperature. Temperatures were recorded hourly using Thermochron iButton dataloggers. The data loggers were suspended in the middle of the tanks, approximately halfway between the surface and the bottom. Temperature differences among tanks were consistent throughout the course of the experiment (Fig S1.1C). Heaters were placed at the bottom of the mesocosms. Mesocosms were covered with two layers of window screen to minimize colonization by other invertebrates. Water levels were maintained by natural precipitation and weekly additions to maintain volume.

**Plankton Sampling and Analysis**

We sampled phytoplankton, chlorophyll *a*, zooplankton, and oxygen concentrations weekly until August 28th, 2012. We sampled algal assemblages in 100-mL water samples collected from ~40-cm below the surface. We counted and identified cells using the Utermöhl sedimentation method [62] and estimated chlorophyll *a* concentration using a Trilogy fluorometer (Turner Designs). Chlorophyll *a* concentration can be used as a proxy for biomass, and though the ratio between chlorophyll a and total biomass can itself vary with temperature, size and species composition [63,64], the chlorophyll *a* concentration represents biomass allocated to photosynthesis and NEP, our measure of ecosystem function.

Phytoplankton were identified and counted to species or taxon level by inverted microscopy. We collected zooplankton samples using a ‘depth integrated zooplankton sampler’. The device is a cylinder 4 cm in diameter and 60 cm in length with a cap at one end. We mixed mesocosm water gently, then submerged vertically the sampler, sealed it, removed it and dumped water in to a bucket. We repeated until we had removed 10 L of water, which was then filtered through a 64-µm to collect zooplankton, and then the filtered water was returned to mesocosms. Plankton was fixed with Lugol’s iodine solution (5%). Under 10x magnification, we counted and identified zooplankton to genus and measured standard length for all development stages in week 8. We measured oxygen concentrations using YSI-85 oxygen sensor (Yellow Springs Instruments, Yellow Springs, Ohio, USA).

**Estimation of biomass and oxygen fluxes**

We estimated whole ecosystem oxygen fluxes using the dissolved oxygen (DO) change technique [65]. Oxygen production during the daytime is the product of photosynthesis minus respiration (net ecosystem production, or NEP), and oxygen depletion during the night is the result of respiration (ER). We compared DO concentrations measured over 24 hours (dawn, dusk and the following dawn). Comparison of oxygen concentrations at dawn, dusk and dawn of the following day (Eqn 5) can indicate not only the cumulative biotic NEP and ER fluxes during that time interval, but also differences in water temperature that affect oxygen concentrations in water. At standard pressure, which is appropriate for our experiment near sea level, oxygen saturation can change by approximately 1 mg/L with a change in temperature of 5 °C, described by:

, Eqn 5

where [O2]water is the O2 concentration of water, [O2]sat is the concentration the water would have if it were at equilibrium with the atmosphere (390 µatm), T is temperature of the observation (°C) [66]. For the differences in temperature we observed, corrections were on the order of mean 0.0002 + sd 0.0008 µmol O2 / L / hr for NPP, and mean 0.0008 + sd 0.0003 µmol O2 / L / hr for ER. Because these values are within 25% of our total observed changes in oxygen during those periods (mean 0.003 + sd 0.001 µmol O2 / L / hr for NEP and mean 0.003 + sd 0.002 µmol O2 / L / hr for ER), we included the correction in our analyses. Overall, the conclusions based on model selection did not depend strongly on the use of the correction (results not shown).

We estimated NEP and ER by converting changes in observed O2 (mg L-1) between daytime observation times (*tdawn*, *tdusk*) and overnight observations (*tdusk*, *tdawn2*) to micromolar concentration (*z* = 31.25 µmol/1 mg), and correcting for changes in estimated equilibrium oxygen concentration () (Eqn 3) due to changes in saturation state with temperature at each time.:

Eqn 6a

. Eqn 6b

**Model and hypothesis development**

The expression of temperature effects on a metabolic rate *bi* – in our case, oxygen production via photosynthesis or consumption via respiration - in this model is a special case of a more complex equation that allows each species to follow a thermal performance curve, often described by a modified Sharpe-Schoolfield equation [18,22,67], in which an individual’s or population’s performance declines at high temperatures above some optimal temperature. We do not use this TPC model here for two reasons: we do not expect photosynthesis or respiration to exceed optimal operating temperatures in our system for most taxa () so we believe the simpler exponential is a suitable hypothesis for cross-system comparison (following Yvon-Durochers, etc etc) [18,22], and we do not have thermal performance data for the many species in our communities that would allow fitting of thermal performance curves within communities.

We modeled PP (eqn 3) this by including a term for trophic level in the intercept term (eqn 3 rearranged and logged):

Eqn 7

We derived the expression for the trophic cascade by relating algal biomass in the AGP and AG treatments:

Eqn 8

We then cancelled terms, and indicated temperature dependence of mass and normalization constants. To approximate their temperature dependence and in the absence of additional information about their functional forms, we used general Arrhenius functions, but we note that other functions could be used if appropriate. Consequently, ratio of these biomasses may vary with temperature according to the relative temperature dependences of thermal traits and size distributions:

Eqn 9

and the strength of the trophic cascade may therefore be expected to decline with a temperature dependence that reflects the temperature dependences of mass and normalized performance for each trophic treatment:

Eqn 10

**Statistical Analysis**

We tested our hypotheses about whether effects of temperature on metabolism are modified at the ecosystem level by species interactions using a regression experimental design involving 30 independent ecosystems (Fig S1.1). For each food chain length (algae-only (A), algae-grazer (AG) or algae-grazer-predator (AGP)), we maintained ecosystems at distinct temperatures in a regression design with mean temperatures ranging from 19.7 (+ 3.15) °C to 26.1 (+ 3.59) °C (Fig S1.1C). The regression design allowed us to estimate slopes (*Ea*, Eqn 1) of response variables along a continuous temperature gradient for different trophic structures by log-transforming equation 1b and fitting linear models to the continuous temperature gradient. We chose the regression design, though unreplicated within temperature levels, because it allowed us to compare activation energies (*Ea,* Eqn 1) fitted over a broad range of temperature; an important test of thermal responses that is not possible with designs with only 2 or even three temperature levels. Regression designs, even without replication within levels, gain statistical power from the range of x-levels tested [68,69].

We used a mixed effects model (lme function in the nlme package of R) to examine the main and interactive effects of temperature (a continuous fixed factor) and food chain length (a categorical fixed factor) on net ecosystem oxygen production, net ecosystem oxygen consumption, and chlorophyll a concentration with a random intercept for individual ecosystems. We used a within-subject mean centering approach to distinguish temperature effects into those associated with an ecosystem’s average temperature over the entire experimental period (a ‘between-ecosystem’ effect) from effects variation in temperature over time (a ‘within-ecosystem temperature’ effect) [70]. The response variable (Y) for each ecosystem *j* in week *i* was modelled as a continuous response to variation in inverted ecosystem temperature (1/kTij) and trophic treatment (*Z*j):

 Eqn 11

where *β0.j(i)* represents an intercept allowed to vary randomly among ecosystems. The terms in the full model (Eqn 5) are: the between-ecosystem effect of temperature (*β2*), estimated as the slope of ln(Y*ij*) on the mean temperature over all weeks for ecosystem *j*, expressed as inverse temperature; the within-ecosystem (*β1*) effect of temperature variation over time estimated as the slope of ln(Y*ij*) *vs* centered weekly temperature; interaction (*β3*) between within-ecosystem temporal variation in temperature and the experimental temperature treatment; trophic species interactions (*β4*), and interactions between species interactions and overall mean (*β5*) and weekly temperature (*β6*).

To test our hypothesis that species interactions modify temperature dependence (*Ea*, Eqn 1) of response variables (Y), we compared models with and without trophic level terms (*β4*)and interactions between *TL* and temperature (*β5*, *β6*). We also tested models without temperature terms for within-system variation (*β4*). In total, the model set included 9 models (Table S1.1). Response variables were ln-transformed prior to analyses to achieve normal distributions and to linearize temperature effects for analysis and to fit *Ea* values from Eqn 1. When modelling, we centered temperature treatment (1/kTj) on the grand mean of all temperatures observations (not shown in Eqn 5) to reduce correlations between slope and intercept terms [71].

To test the effect of temperature on trophic cascade strength, we used the following statistical model:

Eqn 12,

In which the effect of temperature on trophic cascade strength in each temperature treatment *j* was modeled for each week *w* and for the temperature of the tanks, with random effects *uj* were assigned for each power treatment (*j*).

We ranked models using Akaike’s Information Criterion weights (using the MuMin package in R), adjusted for small sample sizes (AICC). When two or more models were considered comparable or equivalent (δAICC < 2) we reported all models meeting this criterion and report averaged coefficients. We estimated *Ea* and intercepts for among-ecosystem responses to temperatures by first rearranging Eqn 12 to group coefficients by temperature term (Eqn 5a).

 Eqn 13

We estimated confidence intervals for composite terms following [72]. We used R statistical software (R v. 1.0.44 R Developmental Core Team 2006)*.* Our models controlled for the effect of temperature variation over time on ecosystem fluxes and biomass within systems.

We determined the effects of temperature and predator presence on zooplankton abundance data using generalized linear mixed effects models with tank as a random effect modeled on a negative binomial regression distribution to account overdispersed Poisson distributed count data (using the glmmADMB package in R).

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