# Quantifying the carbon benefits of ending bottom trawling

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ARISING FROM E. Sala et al. *Nature* https://doi.org/10.1038/s41586-021-03371-z (2021)

Bottom trawling disrupts natural carbon flows in seabed ecosystems owing to sediment mixing, resuspension and changes in the biological community. Sala et al.[1] suggest that seafloor disturbance by industrial trawlers and dredgers results in 0.58–1.47 petagrams (Pg) of aqueous $CO_2$ release annually (equivalent to 0.16–0.4 Pg carbon (C) per year), owing to increased organic carbon (OC) mineralization, which occurs after trawling. We are concerned, however, that Sala et al.[1] overestimate trawl-induced $CO_2$ release, because their model uses a reactivity value ($k$, the first-order decay rate) estimated for highly reactive OC delivered recently to the sediment surface, and apply it to bulk sediment (typically composed of labile, recalcitrant and refractory C), which is known to have a much lower reactivity[2]. These assumptions result in an upward bias in the estimated $CO_2$ release by several orders of magnitude, overestimating the impact of trawling on global OC mineralization rates.

The parameter values in Sala et al.[1] ignore the important role of composition in driving OC mineralization in marine sediments. OC that reaches the sediment represents a mixture of compounds that range from highly reactive to very unreactive molecules[3]. Typically, around 70% (represented by the fraction of reactive material, $p$, of 0.70 for muddy sediment in the model of Sala et al.[1]) is highly reactive and mineralized by microorganisms within the first few centimetres of sediment, which translates into a high $k$ value (reactivity of the OC pool, 1–10 per year ($yr^{-1}$)). The remaining, less-reactive fractions are mineralized at a much slower rate, with typical $k$ values below 0.1 $yr^{-1}$ (ref. 4). Because of the preferential mineralization of the more-reactive fractions, the $k$ value of the bulk OC decreases exponentially with sediment depth, generally from 1–10 $yr^{-1}$ at the sediment–water interface to less than 0.01 $yr^{-1}$ below a depth[4,5] of 5 cm (Fig. 1). The standing stock of OC in the sediment thus typically exhibits a $k$ value of 0.01–0.1 $yr^{-1}$. Consequently, the approach Sala et al.[1] have taken—using a $k$ value of 0.3–17 $yr^{-1}$ and applying this to the bulk of the OC stock—may result in an overestimation of $CO_2$ release of historically buried OC by two to three orders of magnitude. We argue that incorporating the role of composition would require lowering the $k$ value to around 0.01 $yr^{-1}$, which is representative of sub-surface sediment[5], and applying it to the bulk of the sediment (fraction of reactive material, $p = 1$) or, alternatively, using the original high $k$ values ($k = 0.3$–$17$ $yr^{-1}$) and applying them to the fraction of reactive material $p$ present in historically buried OC ($p = 0.001$–$0.01$). More importantly, the calculations in Sala et al.[1] would have given only an estimate of OC remineralization independent of trawling—because these $k$ and $p$ values are representative of OC mineralization in marine sediments (Fig. 1 shows typical $k$ values relative to sediment depth for a range of North Sea sediments).

Furthermore, the OC model presented by Sala et al.[1] does not differentiate between OC mineralization in undisturbed sediments and that induced by sediment disturbance. Instead, Sala et al.[1] implicitly assume that the OC mineralization rate calculated using their model results from trawling disturbance alone. As a result, their model assumptions imply that the OC in an area protected from trawling is unreactive and will not be mineralized. The 'carbon model validation' section in the methods of Sala et al.[1] clearly illustrates this issue. Sala et al.[1] compare the modelled $CO_2$ emissions that derive from only the trawl disturbance of historically buried OC with empirical estimates of $CO_2$ emissions from natural-plus-trawling mineralization of all sedimentary OC, and also do not compare the emissions with those of untrawled control sites. These fundamentally incomparable measures are not suitable for the validation of their model. The fact that these measures are of the same order of magnitude illustrates that $CO_2$ emissions by trawling are likely to be small compared with the emissions from natural mineralization[6] and much smaller than those modelled by Sala et al.[1].

The ultimate question is whether the reactivity of the OC stock is increased by trawling disturbance and resuspension, and thus whether the $k$ value is higher after trawling. Unfortunately, this question is not adequately addressed by Sala et al.[1]. To date, our knowledge of the effects of the disturbance and resuspension of sediments induced by trawling on the reactivity of OC, and how this compares with the effects of natural resuspension events (such as storms and waves) is extremely limited. A recent review of 49 studies investigating OC stocks after trawling-induced disturbances revealed highly mixed results, with 61% of studies reporting no significant effect, 29% reporting lower OC stocks and 10% reporting higher stocks[6]. To robustly estimate the global impact of bottom trawling on OC mineralization, new experiments are needed that quantify the reactivity of disturbed OC in the sediment and in resuspension.

In conclusion, we currently do not know enough about the impact of trawling on seabed carbon to make robust global projections. Reliable estimates of sediment carbon loss should be based on models that use parameter estimates for the change in OC reactivity and that are tested against empirical measurements. Sala et al.[1] suggest that reducing $CO_2$ release through reducing trawling effort could generate carbon credits and provide an opportunity for financing marine protected areas. Although this is certainly an idea worth considering, we argue that the $CO_2$ release estimates of Sala et al.[1] create unrealistic expectations about

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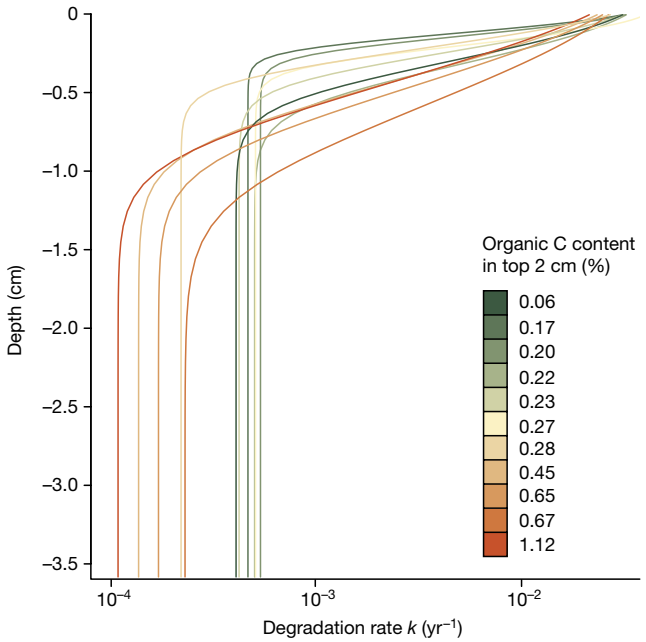


**Fig. 1 | Decrease in modelled OC degradation-rate constants with sediment depth for 11 sites in the North Sea, with varying OC contents at the sediment surface.** The mean rates stem from the degradation of OC, which consist of a reactive and a less-reactive OC fraction. Each of these fractions has a different degradation rate *k*. Data and modelling results are from a previously published study[7].

the quantity of carbon credits that can be generated. Even initial plans for the management of bottom trawling for carbon benefits require estimates that are of the correct order of magnitude, and we think that the study by Sala et al.[1] does not supply those estimates.

## Reporting summary

Further information on research design is available in the Nature Portfolio Reporting Summary linked to this article.

## Online content

Any methods, additional references, Nature Portfolio reporting summaries, source data, extended data, supplementary information, acknowledgements, peer review information; details of author contributions and competing interests; and statements of data and code availability are available at https://doi.org/10.1038/s41586-023-06014-7.


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**Author contributions** J.G.H., S.J.v.d.V., R.A.M., E.D.B., J.T., M.J.K., A.K.S. and M.S. all contributed to the ideas and writing of the paper. J.G.H. wrote the initial draft and S.J.v.d.V., R.A.M., E.D.B., J.T., M.J.K., A.K.S. and M.S. contributed to refining and editing the text.

**Competing interests** M.J.K. is currently funded by the Natural Environmental Research Council (UKRI-GCRF), EDP Renewables, Fishmongers Company, Macduff Shellfish, Scottish Whitefish Producers Organisation, Sainsbury's Supermarkets, The Cooperative, CISCO and the Defra Seafood Innovation Fund. Less than 1% of his current funding is from fishing organizations. M.S. has received funding from the Seafood Innovation Fund, which funds collaborative industry–science projects, although no funding is received directly from the fishing industry. The other authors declare no competing interests.

**Additional information**


**Supplementary information** The online version contains supplementary material available at https://doi.org/10.1038/s41586-023-06014-7.



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|---|---|
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| | |
|---|---|
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| Research sample | No samples were taken |
| Sampling strategy | No samples were taken |
| Data collection | No data was collected |
| Timing and spatial scale | No data was collected |
| Data exclusions | No data was collected |
| Reproducibility | No data was collected |
| Randomization | No data was collected |
| Blinding | No data was collected |

Did the study involve field work? ☐ Yes ☒ No

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## Methods

| n/a | Involved in the study |
|---|---|
| ☒ | ☐ ChIP-seq |
| ☒ | ☐ Flow cytometry |
| ☒ | ☐ MRI-based neuroimaging |

# Reply to: Quantifying the carbon benefits of ending bottom trawling

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REPLYING TO: J. G. Hiddink et al. *Nature* https://doi.org/10.1038/s41586-023-06014-7 (2023)

In the accompanying Comment, Hiddink et al.[1] challenge our estimate[2] of the magnitude of seabed carbon remineralized by bottom trawling. However, we think that the conclusions by Hiddink[1] are based on incorrect assumptions and that these conclusions lack quantitative support for several of their claims.

We think that the assertion by Hiddink et al.[1] that our calculations suggest that "OC in an area protected from trawling is unreactive and will not be mineralized" is incorrect. The model by Sala et al.[2] rests on the idea that trawling exposes carbon buried in less-active sediment layers to shallower, more biogeochemically active ones. Because some of the affected sediment is eroded and transported away by ocean currents[3], each trawling event exposes subsequent deeper layers of buried carbon. As a result, trawling aids in decomposing previously buried carbon by acting as a mechanism that can reoxygenate surface sediments, transport previously buried carbon to more biologically active zones and release carbon from physical occlusion.

Decades of research have shown that oxygen and oxygen exposure time have essential roles in the degradation of marine organic matter, especially older organic carbon and organic carbon associated with mineral surfaces[4,5]. Bottom trawling exposes previously buried carbon to more oxygenated environments because, as fishing gear scrapes the seafloor, it mixes and resuspends sediments and their associated carbon[3]. Although most of the research on the impacts of oxygen on carbon degradation focuses on long-term exposure, some studies suggest that oxygen exposure over a few days to weeks can increase carbon turnover in marine sediments[6]. Studies on trawling have shown that sediment plumes take several days to dissipate, and trawling tracks in the sediment can persist for months to years[3]. Furthermore, lateral transport of resuspended carbon and frequent trawling of sites over the past 30–70 years can increase the duration and frequency of oxygen exposure[3].

Hiddink et al.[1] further suggest that our selection of $k$ values (first-order decay rates) overestimates trawling-induced $CO_2$ efflux by two to three orders of magnitude. It is important to understand that $k$ is derived and modelled using indirect methods and cannot be measured directly. Like all models, the theoretical assumptions underpinning the carbon model can affect its predictions, and we acknowledge that $k$ is a crucial parameter that needs to be well constrained to give accurate predictions. Here we discuss and compare the approach suggested by Hiddink et al.[1] with models described in the literature and with our own empirical synthesis.

The approach by Hiddink et al.[1] is based on the paradigm that the degradation of organic matter is controlled by only its chemical properties, leading to predictable declines in $k$ values with depth. However, several authors of a previously published paper[7], which is extensively cited by Hiddink et al.[1] in their discussion of $k$ values, argue in their recent publication[8] that "the relative importance of organic structure and composition in controlling overall preservation/degradation remains unclear". Research at the forefront of marine organic-matter preservation now focuses on the idea that the interplay between the biological, geochemical and physical attributes of the environment govern organic-matter reactivity[8–10]. As a result of this paradigm shift, recent studies have argued that the fixed reactivity paradigm used by Hiddink et al.[1] is no longer valid, because it generates misleading conclusions about carbon stability and reactivity[9].

By contrast, in our previous paper[2], we determined empirical $k$ values using a broad literature search to constrain the range of possible $k$ values to between 0.275 and 16.8 (not 1–17 $yr^{-1}$, as suggested by Hiddink et al.[1]). We then used four independent, trawled sites with measured $CO_2$ effluxes to characterize and validate the $k$ values and the proportion of available carbon across different regions and sediment types. It is our view that deriving $k$ values from an analysis of field data, as was done in Sala et al.[2], is a more robust approach than the suggestion by Hiddink et al.[1] of a single global $k$ value in the range of 0.01–0.1, which was selected using a theoretical approach that can be subjected to expert biases.

To analyse the assumption by Hiddink et al.[1], we followed the same validation methods as described previously[2], except that we used $k$ values for our validation sites that were one or two orders of magnitude lower than in our original model[2]. We found that the percentage of error of the $CO_2$ efflux increased significantly from a mean of 21% (19–24%), when using the original $k$ values[2], to a mean underestimate of 63%

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**Table 1 | Model comparisons of trawled validation sites using different *k* values and natural fluxes to estimate $CO_2$ efflux**

| | Original model | | *k* reduced by one order of magnitude | | *k* reduced by two orders of magnitude | |
|---|---|---|---|---|---|---|
| **Site** | ***k*** **($yr^{-1}$)** | **Error (%)** | ***k*** **($yr^{-1}$)** | **Error (%)** | ***k*** **($yr^{-1}$)** | **Error (%)** |
| **Approach 1** | | | | | | |
| Westerschelde | 1.0 | 4 (0.7 to 6) | 0.1 | −79 (−76 to −82) | 0.01 | −92 (−89 to −95) |
| Thermaic Gulf | 12.3 | 8 (−3 to 18) | 1.23 | −17 (−7 to −27) | 0.12 | −67 (−56 to −77) |
| Sweden | 1.0 | 39 (19 to 59) | 0.1 | −39 (−19 to −59) | 0.01 | −51 (−31 to −72) |
| Bay of Aarhus | 1.0 | −45 (−38 to −52) | 0.1 | −78 (−71 to −85) | 0.01 | −83 (−75 to −90) |
| **Approach 2** | | | | | | |
| Westerschelde | 1.0 | −2 | 0.1 | −86 | 0.01 | −98 |
| Thermaic Gulf | 12.3 | −13 | 1.23 | −37 | 0.12 | −87 |
| Sweden | 1.0 | 0.2 | 0.1 | −78 | 0.01 | −90 |
| Bay of Aarhus | 1.0 | −59 | 0.1 | −91 | 0.01 | −96 |

Approach 1 includes *k* values and trawling flux data from Sala et al.[2]; the natural flux is 19 Mg $CO_2$ $km^{-2}$ $yr^{-1}$, as estimated previously[11]. Natural $CO_2$ fluxes[11] represent the global mean (lower and upper bounds) of fluxes for water depths between 50 m and 2,000 m. Approach 2 includes *k* values and trawling flux data from Sala et al.[2]; the natural flux is 16 Mg $CO^2$ $km^{-2}$ $yr^{-1}$, as estimated previously[12]. Natural $CO_2$ fluxes[12] represent the region-specific mean. Annual natural $CO_2$ fluxes represent 90% of the annual carbon flux to sediments. The combined flux is the sum of the trawling and natural fluxes. The mean error (%) is the difference between the model predictions (trawled flux + natural flux) and the measured flux. Additional information for the calculation of the error percentages can be found in Supplementary Table 1. The full methodology for validations has been described previously[2].

(58–68%) when *k* was reduced by one order of magnitude, and of 83% (78–88%) when *k* was decreased by two orders of magnitude (Table 1). As a result, the suggestion by Hiddink et al.[1] of using a *k* value in the range of 0.01–0.1 leaves the carbon equation unbalanced, with a considerable source of remineralized carbon that is not accounted for and that the annual supply of fresh carbon cannot explain. We acknowledge that using mean global and regional natural fluxes does not account for the considerable variation often observed between sites. However, to make up for the deficits in the carbon budgets, site-specific mean carbon fluxes to the seafloor would have to be 2–15 times greater than the mean global average[11] and 10–86 times greater than regional averages[12].

Furthermore, our data do not support the claim by Hiddink et al.[1] that using higher *k* values overestimated $CO_2$ efflux by two to three orders of magnitude. Using the carbon benefit methodology from Sala et al.[2], we reduced all *k* values by one and two orders of magnitude by dividing the original *k* values by 10 and 100, respectively. We then examined the validity of the claim by Hiddink et al.[1] that such reductions in *k* values would result in a two-to-three order-of-magnitude lower $CO_2$ efflux. Decreasing the *k* values by one order of magnitude (global mean *k* value = 0.26 $yr^{-1}$) resulted in a trawling-induced efflux of 0.35 Pg $yr^{-1}$, which is in the same order of magnitude as the lower bound estimate of 0.58 Pg $yr^{-1}$ by Sala et al.[2] (Table 2). Reducing the *k* values by two orders of magnitude (global mean *k* value = 0.026 $yr^{-1}$) reduced the global efflux to 0.043 Pg $yr^{-1}$, only one order of magnitude lower than the lowest estimate by Sala et al.[2] (Table 2), not the two to three orders of magnitude suggested by Hiddink et al.[1].

Since the publication of our previous paper[2], another study[13] has synthesized the literature examining trawling impacts on carbon with a mostly qualitative review. Although this study[13] found mixed results for the effects of trawling on carbon stocks, the only two studies that examined remineralization and/or metabolism at the same site before and after trawling found that trawling increased remineralization or metabolism. Furthermore, the study[13] found that most papers that reported no effect of trawling on carbon stocks had three commonalities: these studies focused on trawling intensity (that is, they lacked untrawled control sites); were conducted in sandy sediments; and focused on shallower sediment depths. Notably, the study[13] acknowledges the potentially considerable effect trawling could have on seabed carbon and highlights the need for more robust studies on this subject, a sentiment we also echo here and in our previous paper[2].

Overall, we have flagged several concerns about the lack of quantitative support for the claims by Hiddink et al.[1] and we question the use of theoretical predictions instead of empirical data from trawled sites; especially because their theoretical approach to *k* values, which lends itself to expert biases, has been recently scrutinized by soil scientists and biogeochemists[8–10]. Although we acknowledge the limitations of estimating global carbon efflux from marine sediments because of the scarcity of data, our model lays the foundations for future work and provides a reasonable estimate to be improved on when better data become available. We welcome further rigorous research that elucidates the impacts of trawling on seabed carbon storage.

## Reporting summary

Further information on experimental design is available in the Nature Portfolio Reporting Summary linked to this Article.

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**Table 2 | Comparison of parameters and the global estimate of trawling-induced $CO_2$ fluxes**

| Parameter | Estimate |
|---|---|
| **Original model** | |
| Global trawling-induced $CO_2$ flux (Pg $yr^{-1}$) | 1.47–0.58 |
| Global mean *k* value ($yr^{-1}$) | 2.6 |
| Global mean percentage of remineralization | 29.70% |
| **The model with one order of magnitude reduction in *k* values** | |
| Global trawling-induced $CO_2$ flux (Pg $yr^{-1}$) | 0.348 |
| Global mean *k* value ($yr^{-1}$) | 0.26 |
| Global mean percentage of remineralization | 6.80% |
| **The model with two orders of magnitude reduction in *k* value** | |
| Global trawling-induced $CO_2$ flux (Pg $yr^{-1}$) | 0.043 |
| Global mean *k* value ($yr^{-1}$) | 0.026 |
| Global mean percentage of remineralization | 1.20% |

Comparisons between the original model in Sala et al.[2] and two further models in which the *k* values were reduced by one or two orders of magnitude. The methods used for model calculations have been published previously[2].

## Data availability

Data for Table 2 can be found in the supplementary information of Sala et al.[2].

## Code availability

R code for Table 2 can be found at GitHub (https://github.com/emlab-ucsb/ocean-conservation-priorities).


1. Hiddink, J. G. et al. Quantifying the carbon benefits of ending bottom trawling. *Nature* https://doi.org/10.1038/s41586-023-06014-7 (2023).
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**Competing interests** The authors declare no competing interests.

**Additional information**
**Supplementary information** The online version contains supplementary material available at https://doi.org/10.1038/s41586-023-06015-6.


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For all statistical analyses, confirm that the following items are present in the figure legend, table legend, main text, or Methods section.

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| ☒ | ☐ | The exact sample size (*n*) for each experimental group/condition, given as a discrete number and unit of measurement |
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| ☒ | ☐ | A description of all covariates tested |
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| ☒ | ☐ | For null hypothesis testing, the test statistic (e.g. *F*, *t*, *r*) with confidence intervals, effect sizes, degrees of freedom and *P* value noted<br>*Give P values as exact values whenever suitable.* |
| ☒ | ☐ | For Bayesian analysis, information on the choice of priors and Markov chain Monte Carlo settings |
| ☒ | ☐ | For hierarchical and complex designs, identification of the appropriate level for tests and full reporting of outcomes |
| ☒ | ☐ | Estimates of effect sizes (e.g. Cohen's *d*, Pearson's *r*), indicating how they were calculated |

*Our web collection on statistics for biologists contains articles on many of the points above.*

## Software and code

Policy information about availability of computer code

| | |
|---|---|
| Data collection | no software was used |
| Data analysis | R code for table 2 can be found at https://github.com/emlab-ucsb/ocean-conservation-priorities. |

For manuscripts utilizing custom algorithms or software that are central to the research but not yet described in published literature, software must be made available to editors and reviewers. We strongly encourage code deposition in a community repository (e.g. GitHub). See the Nature Portfolio guidelines for submitting code & software for further information.

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Data for table 2 can be found in the supplementary material of Sala et al. (2021) https://doi.org/10.1038/s41586-021-03371-z

## Human research participants

Policy information about studies involving human research participants and Sex and Gender in Research.

| | |
|---|---|
| Reporting on sex and gender | No human research participants were used in this study |
| Population characteristics | No human research participants were used in this study |
| Recruitment | No human research participants were used in this study |
| Ethics oversight | No human research participants were used in this study |

Note that full information on the approval of the study protocol must also be provided in the manuscript.

# Field-specific reporting

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# Ecological, evolutionary & environmental sciences study design

All studies must disclose on these points even when the disclosure is negative.

| | |
|---|---|
| Study description | This manuscript is a Reply to Hiddink et al.'s "Quantifying the carbon benefits of ending bottom trawling"., which was submitted as a Matter's Arising to Sala et al. 2021 https://doi.org/10.1038/s41586-021-03371-z. It therefore has no |
| Research sample | No samples were collected for this Reply and all data were obtained from the original article Sala et al. (2021) https://doi.org/10.1038/s41586-021-03371-z |
| Sampling strategy | No samples were collected for this Reply and all data were obtained from the original article Sala et al. 2021 https://doi.org/10.1038/s41586-021-03371-z |
| Data collection | All data were obtained from the original article Sala et al. (2021) https://doi.org/10.1038/s41586-021-03371-z by TBA and JM |
| Timing and spatial scale | *Indicate the start and stop dates of data collection, noting the frequency and periodicity of sampling and providing a rationale for these choices. If there is a gap between collection periods, state the dates for each sample cohort. Specify the spatial scale from which the data are taken* |
| Data exclusions | No data were excluded from the study |
| Reproducibility | No samples were collected for this Reply and there were no experiments. All data and code can be found were obtained from the original article Sala et al. (2021) https://doi.org/10.1038/s41586-021-03371-z |
| Randomization | No samples were collected for this Reply |
| Blinding | No samples were collected for this Reply |

Did the study involve field work? ☐ Yes ☒ No

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We require information from authors about some types of materials, experimental systems and methods used in many studies. Here, indicate whether each material, system or method listed is relevant to your study. If you are not sure if a list item applies to your research, read the appropriate section before selecting a response.

## Materials & experimental systems

| n/a | Involved in the study |
|---|---|
| ☒ | ☐ Antibodies |
| ☒ | ☐ Eukaryotic cell lines |
| ☒ | ☐ Palaeontology and archaeology |
| ☒ | ☐ Animals and other organisms |
| ☒ | ☐ Clinical data |
| ☒ | ☐ Dual use research of concern |

## Methods

| n/a | Involved in the study |
|---|---|
| ☒ | ☐ ChIP-seq |
| ☒ | ☐ Flow cytometry |
| ☒ | ☐ MRI-based neuroimaging |