Dear reviewers, thank you for your valuable feedback. We have taken into account the insightful suggestions provided, which have allowed us to elaborate further on the content and improve the overall quality of the paper. The responses to the comments are presented below in a two-column format.

Comments	**Response**
Comments on how the overall study supports the conclusions presented in the title or discussion/conclusion sections. The title itself suggests that the paper may include a framework that addresses the integrated economic-environmental viability of dams. Indeed, this is a very important issue. However, the paper itself has no actual definition of what a "viable" set of dams is, how the scenarios compared differ in terms of their "viability". The title is therefore misleading. The actual scope is essentially a trade-off analysis between dam implementation scenarios for some of the 4 objectives considered, without offering a critical analysis of the viability of the resulting metrics across scenarios, and simply focusing on comparing values. Is any of the configurations shown in the scenarios economically-environmentally viable, and if so, for whom? In order to coherently approach the statement in the title, I consider that the authors need to provide: A definition of what is a viable configuration of dams. There are several possibilities. For example, it could be based on stakeholder preferences and/or, from a purely phenomenological perspective, the identification of boundaries or tipping points in the system that may preclude the functioning of a key component, e.g., fish biodiversity, agriculture, energy production, etc	The definition of the viability of dams has been provided, along with an explanation to justify it with the present study. The paper has been updated with the following content to address the combined economic-environmental viability of the dams. The following paragraph is added to the revised version of the paper: "A viable configuration of dams refers to a set of parameters and characteristics that consider various factors such as stakeholder preferences and ecosystem preservation to ensure the sustainable and optimal functioning of a dam system. From a stakeholder perspective, it takes into account the preferences and needs of different parties involved, including local communities, government bodies, environmental organizations, and industries. The aim is to strike a balance among diverse interests, incorporating stakeholder preferences into the design and operation of the dam system (Kemmler & Spreng, 2007). From a phenomenological perspective, a viable configuration respects the boundaries within the ecosystem that, if exceeded, could disrupt the functioning of key components such as fish biodiversity, aquatic habitats, and downstream water quality (Kumar and Katoch, 2014). Overall, achieving a sustainable balance between societal needs and environmental protection requires careful planning, scientific analysis, and transparent decision-making processes in dam development (Kemmler & Spreng, 2007; Kumar and Katoch, 2014). In this study, we have chosen agriculture production and fish species richness as indicators to represent both economic development and environmental sustainability. By selecting these components, we aim to find a balance that considers the needs of society while also ensuring the protection of the environment by identifying a specific configuration from a collection of existing dams"

Evidence that the proposed components and indicators are relevant in the context of the proposed case study stakeholders, and that these indicators can be quantified with a reasonable degree of certainty. It is not clear in the paper why the 4 selected indicators were chosen and whether they are representative of system processes or priorities. See comment section 2 for some details.	The selected components and indicators for this study have been justified based on their relevance in assessing the integrated economic and environmental viability of dams. To ensure a comprehensive analysis, two indicators that represent key ecosystem priorities, namely agriculture production and fish species richness, have been chosen. These indicators are considered essential in understanding the impact of the flow regime on the ecosystem. To quantify these indicators with a reasonable degree of certainty, two separate formulas have been employed. The following paragraph is added to the revised version of the paper: "Ecosystems have the capacity to provide multiple services simultaneously, but it is generally challenging to manage them in a way that maximizes all services at once (King et al., 2015; Bennett et al., 2009). This can result in trade-offs, where prioritizing one service may come at the expense of others. In river basins, trade-offs often occur as a consequence of management decisions, leading to conflicts between upstream and downstream users. An example is the trade-off between agricultural yield and downstream water quality (Stosch et al.,2019). Another study has examined the cost-effectiveness of hydropower production economics versus salmon habitat restoration costs in relation to the productivity of Atlantic salmon (Bustos et al., 2017). In the Cauvery basin, approximately 48 percent of the land is used for cultivation (Singh, 2013). In certain stretches of the Cauvery River, there is extensive water abstraction for intensive agriculture (Vedula, 1985; Bhave et al. 2018). This water extraction has resulted in notable changes in the composition of aquatic species, primarily due to the construction of reservoirs. These alterations have had an impact on the overall biodiversity of the river ecosystem. This results in tradeoff between agricultural production and fish species richness. Therefore, these indicators are deemed appropriate for the study."

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<td>A revised analysis of the interaction between environmental and economic objectives. The current Pareto production frontier analysis only considers the economic component from the perspective of the value of crops, leaving out the monetary value of fisheries, energy, etc.</td>
<td>A partial tradeoff analysis has been conducted, focusing on selected indicators or proxies, while excluding the monetary value of fisheries and energy. This approach allows for a more targeted evaluation of specific factors without incorporating the financial aspects associated with fisheries and energy production.
A comprehensive justification backed up by references has been provided and is included in the paper as follows:
“Cavender-Bares et al. (2015) has demonstrated that for ecological processes and renewable resources, the frontier represents the equilibrium solution derived from a system of equations that represent ecological interactions influencing the yield/synthetic metric/proxy of the two services under consideration. It would also be possible for the axes to represent bundles of services that are assessed using a synthetic metric, indicator, or proxy. The model does not necessitate monetary valuations of ecosystem services. Any quantitative measure of an ecosystem service can be plotted versus another, based on the theoretical or empirically observed relationship governing their joint production (Cavender-Bares et al., 2015). Although the present analysis does not account for the capture of riverine and culture fisheries in reservoirs, it is estimated that the economic value of fisheries is approximately $0.59 million per year, which is about 12 percent of the economic value of agricultural production ($5 million per year). Additionally, the analysis does not include electricity generation from only one reservoir as rest of the reservoirs are used for irrigation. Nevertheless, it is recognized that a complete valuation that considers these aspects is imperative, as emphasized in the discussion section.”</td>
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<td colspan="2">Comments on the specific methodologies used to evaluate each of the proposed environmental or economic components of the proposed framework.
In its current form, the paper describes too succinctly many key components of the analytical framework presented in Figure 1, leaving important gaps in the justification of the selected metrics and the methods used to quantify them. It is understandable that, given the conceptual scope of the work, simplification in the main text may be necessary. However, supplementary materials are required to provide additional details that ensure reproducibility and clarity.
Similarly, there appear to be some important limitations in the proposed analytical components. The four most prevalent are as follows:</td>
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Comments	Response
• **Hydrological modelling:** It is not clear whether the "landscape hydrological model" is essentially the same as that presented in Ekka et al. (2022), or whether new calibrations were performed for this paper.	The integrated modelling is the same as that presented in Ekka et al. (2022) and that it is used here to simulate runoff at the most downstream gauging stations for various configurations of the reservoirs, where corresponding reservoir models are added or removed from the basin wide FLEX-topo models in a plug and play manner. No new calibration was therefore performed in this paper.
• In any case, the reported performance of the model is relatively low (NSE criteria are considered acceptable in the range of +0.2 to +0.5, and good above +0.5. The reported scores are all negative). There is also no mention of the modeling period (assumed to be the same as in Ekka et al., 2022, is 3 years?) Low performance levels can significantly affect the ability to implement data intensive methods such as IHA.	Kindly note that we reported negative (-NSE), since -NSE and MAE were used as objectives of the multi objective optimization algorithm (that finds pareto frontier minimizing these two objectives simultaneously) used to calibrate the integrated model (comprised of Flex-Topo and reservoir model) before its used to simulate stream flows for various reservoir configurations. That means, the negative reverse of NSE value is used to calibrate and validate the model parameters. The -NSE of -0.5 and -0.7 for various calibration and validations steps were reported, which means NSE > 0.5, reasonably good performance. We now explain our response in greater detail and this has been added in the paper as follows: "To calibrate and validate the FLEX-Topo models in Ekka et al. (2022), the dataset of topographic maps, rainfall, and potential evapotranspiration was used. Specifically, the dataset from January 1991 to December 2010 was used for calibration, and the dataset from 2010 to 2016 was used for validation. The reservoir models were calibrated using the dataset composed of inflow, outflow, storage, rainfall, and potential evapotranspiration, for the reservoirs covering the period from January 2011 to December 2016. The Elitist Non-Dominated Sorting Genetic (NSGA-II) algorithm was used to calibrate the model parameters (Deb et al., 2000). Two objective functions are defined and minimized

simultaneously. The first objective (f_1) is the negative of Nash-Sutcliffe Efficiency (NSE) and the second objective (f_2) is the Mean Absolute Error (MAE).

$$f_1 = -NSE = -1 + \frac{\sum_{i=1}^{n}\left(Q_i^m - Q_i^o\right)^2}{\sum_{i=1}^{n}\left(Q_i^o - \bar{Q}_o\right)^2}$$

$$f_2 = \text{MAE} = \frac{1}{n} \sum_{i=1}^{n} \left|Q_i^o - Q_i^m\right|$$

Here, Q_i^m is the i^{th} observation for the observed discharge being evaluated. Q_i^o is the i^{th} value of the modelled discharge. $\bar{Q}_o$ is the mean of observed discharge and *n* being the total number of observations. The parameter sets calibrated for the FLEX-Topo model and the reservoir model are provided in supplementary materials. The NSGA-II parameter setting may have different impacts on computational effectiveness. The results of the calibration and validation after integration of all the reservoirs and measured at the last gauge station is presented in Table 1.

Table 1. The calibration and validation after integration of all the reservoirs

Model performance	**-NSE [range]**	**MAE [range] (10^6 m^3 day^{-1})**
Reservoir Calibration (2011-2016)	-0.68 [-0.67 - (-0.69)]	0.71 [0.70 - 0.72]
Flex-Topo Calibration (1991-2010)	-0.53 [-0.54 - (-0.52)]	0.92 [0.92 -0.97]
Flex-Topo Validation (2011-2016)	-0.50	0.86

	Within parentheses, the Pareto front ranges produced by the NSGA II algorithm are given for both -NSE and MAE. The MAE is always non-negative, and a lower value means a better prediction. The MAE value was recorded in the range of 0.70 - 0.72 (10^6 m^3 day^{-1}) which is in the acceptable range. Similarly, the -NSE value was observed between -0.51 to -0.73. The -NSE value less than -0.50 is acceptable. Kindly note our response to the misunderstanding about our reporting of NSE. We had reported negative NSE values instead of positive NSE (due to its use to maximize as an objective in NSGA II). NSE values are greater than 0.5 and sometimes even around 0.7, which indicates reasonably good performance of a model at daily scale that also incorporated reservoir operations. In the current model, the negative reverse of NSE value is used to calibrate and validate the model parameters"
IHA methodology application is not sufficiently justifies or documented: In the case of freshwater habitat alterations, the paper selected a subset of IHAs. Why the IHA approach and why a subset of IHA indicators?	The IHA technique is sufficiently elaborated in the revised manuscript as follows: "The Indicators of Hydrological Alteration (IHA), initially proposed by Richter et al. (1996), are used to measure the effects of different reservoir combinations on the flow regime in the Upper Cauvery basin. These indicators consider parameters that have significant relationships with river ecosystems, making them suitable for assessing the impact of dams, barrages, and other water diversion structures on the flow regime. While some other methods of assessing the impact of impoundments on river channels involved field surveys, statistical analyses (Yan, 2010), and geomorphic change detection tools (Wheaton, 2015), the Range of Variability Approach and the associated IHA framework provide a more systematic assessment of flow changes. The IHA method utilizes daily streamflow values and characterizes a flow regime based on factors such as magnitude, duration, frequency, timing, and rate of change of flows. Although the application of the IHA method has been relatively limited in studies of Indian rivers (Mittal et al., 2014,

	Kumar and Jayakumar, 2020, Borgohain et al., 2019), this study aims to use the IHA method to gain valuable insights into the impacts of major dams on the flow regime of the Upper Cauvery basin. By doing so, it contributes to a better understanding of the ecological consequences of water diversion and reservoir operations in the region. Major IHA indicators based on their ecological relevance and their ability to reflect human-induced changes in flow regimes which directly impacts the five groups of hydrological features, that is, flow magnitude, duration, timing, frequency and rate of change are used for the present study".
Why not consider other aspects of physical habitat change such as fragmentation, sediment trapping, etc.?	Additional aspects of physical habitat change such as fragmentation, sediment trapping was beyond the scope of the study. However, these additional aspects would be considered in future studies.
Also, can the IHAs be calculated with a reasonable level of confidence given the significant margin of error in several flow components of the hydrologic model?	Yes, IHAs can be calculated with a reasonable level of confidence and this has been explained and added in the paper as follows: "The IHA is typically calculated based on various flow components derived from hydrological models or observed data. While it is true that there can be uncertainties and margin of error associated with flow components in hydrological models, it is possible to calculate the IHA with a reasonable level of confidence, provided that the uncertainties are appropriately addressed. In the present case, the hydrological model is calibrated and validated to ensure that it adequately represents the real-world hydrological processes and assesses its performance."
• **Fish species richness is not an ecosystem service.** It is a metric of biodiversity on evolutionary timescales (i.e., how biophysical processes over thousands to millions of years have produced a particular assemblage of species in a region). More importantly, it	Indeed, fish species richness is not an ecosystem service, however, it serves as a measure of biodiversity and play an important role in upholding the river ecosystem health. And therefore, it is crucial to see fisheries from an environmental sustainability point of view.

does not necessarily explain provision services such as fisheries productivity (for example, aquaculture in reservoirs typically has very high productivity with very low biodiversity).	We now explain our response in greater detail and this has been added in the paper as follows: “Fish species richness refers to the number of distinct fish species found in a specific environment or ecosystem. Although it is not considered a direct ecosystem service, it serves as a measure of biodiversity and reflects the diversity of fish species within a given habitat. Biodiversity, including fish species richness, holds significant importance in ecosystems and possesses inherent value. Fisheries productivity, on the other hand, refers to an ecosystem's capacity to support fish populations that can be harvested for sustenance or other purposes. High species richness can potentially contribute to increased fisheries productivity, as diverse ecosystems often exhibit a range of ecological interactions that sustain robust fish populations. While the significance of fisheries for human welfare is primarily focused on livelihood generation, food production, and nutritional security, freshwater fisheries offer more than just a source of sustenance and livelihoods (Pownkumar et al., 2022). Fish populations play a vital role in upholding the health of river ecosystems, thereby contributing to the sustainability and resilience of rivers (Holmlund & Hammer, 1999). Unfortunately, the broader role of fisheries is often overlooked in ecosystem management decisions, particularly when freshwater resources are allocated for competing purposes such as irrigation, hydropower, and domestic/industrial use. Therefore, it is essential to see fisheries from an environmental sustainability point of view. While we acknowledge that fish productivity should also be considered alongside agriculture production in economic valuation, it is currently 10 % of the value of agriculture production and therefore is left for consideration in future studies”
The estimation of fish species richness was based on a global statistical model developed with the purpose of explaining the global distribution of biodiversity, but NOT of predicting changes in biodiversity based	We agree that the estimation of fish species richness was based on a global statistical model developed by Iwasaki et al. (2012). However, the same formula is validated in 84 major basins

on short term changes in flows. Also, the cited model was developed based on global datasets with no source data in areas such as the case study, and no reference to validation is made. The proposed model is NOT appropriate for this study, as it suggests that by increasing the mean flows over a few years, you'd Therefore, the adoption of the Fish Species richness and model based on flow as a predictor of freshwater ecosystem services is not adequate for the analytical purposes stated in the paper and must be revised.	worldwide by Yoshikawa et al. (2014). Therefore, this methodology adopted is adequate for studying the fish species richness in the basin. We now explain our response in greater detail and this has been added in the paper as follows: "The FSR (Fish Species Richness) value is derived based on a global statistical model developed by Iwasaki et al. (2012). And the model is being validated in 84 major basins worldwide by Yoshikawa et al. (2014). The value obtained from the equation presented by Iwasaki et al. (2012) is centred in the 20-250 range. Other field studies studies have confirmed that the FSR in cauvery river basin tends to be 146 fish species belonging to 52 families (Koushlesh et al., 2021). In the current study, the estimated FSR for the Cauvery River basin ranges from 70 to 123 species under different projected scenarios. This range of values provides sufficient validation for the results obtained" Kindly note that the primary objective of using FSR is not to predict FSR values for the basin, but rather to demonstrate how the characteristics of the river basin and its flow can impact fish species richness and different choices of the configuration of the reservoirs can lead to be different economic values and (fish) biodiversity in the long run (since we are using averages of these two variables over 16 years). This, in turn, affects the overall biodiversity of the river ecosystem and subsequently leads to a decline in river sustainability and resilience. By assessing and understanding these relationships, it becomes possible to identify the potential impacts of flow alterations and basin modifications on the long-run biodiversity and ecological stability of the river systems"
The Production Possibility Frontier generalizes far beyond the data point ranges. It is not clear how the authors arrived at the shape of the PPF given the sparse data points of the model output. Also, the	The shape of the PPF was to mimic the convex hull of points in the tradeoff space that correspond to the 16 reservoir configurations. We acknowledge that the PPF is based on a partial tradeoff

PPF is based on a partial analysis of the monetary value of the system's production and is therefore not representative of the production possibilities of the basin, but only of one sector	analysis between ecosystem services and do not represent the PPF of the whole basin. We give a detailed explanation as follows: "The shape of the PPF was to mimic the convex hull of points in the tradeoff space that correspond to the 16 reservoir configurations. Since we limited our analysis to the existing set of reservoirs (and did not synthetically include any new reservoirs, which might have provided us with more exhaustive set of points, but this would have been more difficult if not impossible to validate), we also limited our conclusion based on this convex hull, i.e. comparative assessment of dominating and non-dominating reservoir combinations in terms of agricultural production and FSR. To clarify this, we have now updated the figure to clearly show the convex hull. The central focus of this paper is to evaluate the tradeoff between dominant services that are governed by river flow regime. That is also why the values of fish production and hydropower generation have not been considered in the present analysis. We recognize that constructing a production possibility frontier for all the ecosystem services of the basin is an extensive task that requires substantial data for the analysis which is beyond the scope of this study."
Besides the major points mentioned above, it is also worth noting that the formal presentation of contextual data and results, such as maps, tables, and graphs, is sometimes redundant. For example, • Table 2 shows the same information as Figures 9 and 10	Table 2 is deleted. The figure 2 is moved to the supplementary materials.

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- The maps/graphs shown in Figures 2, 3, 4, and 5 could be combined into a single figure.

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However, figure 3 is modified based on query from another reviewer but was impossible to merge with Figure 4. We therefore had to keep both.

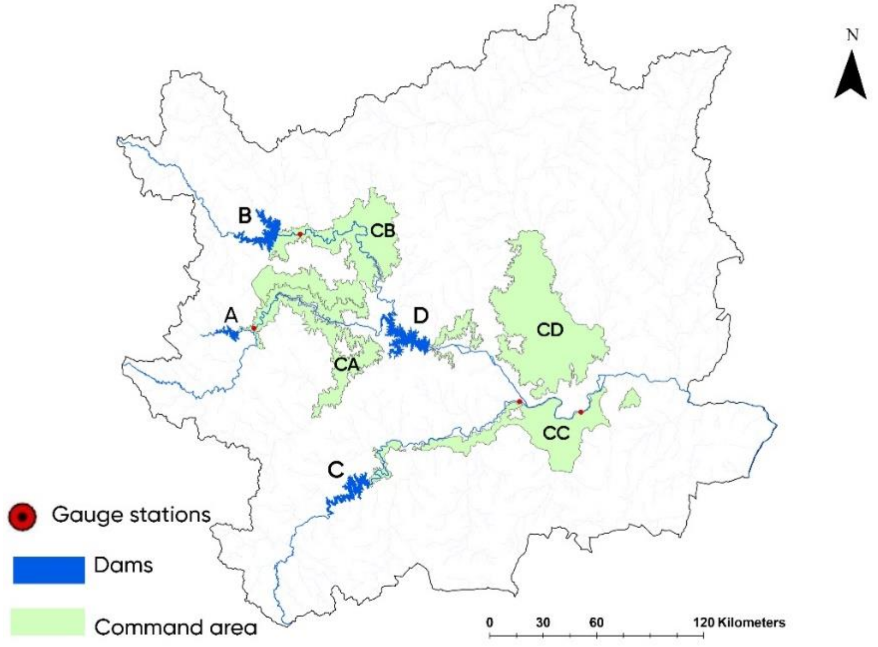

Figure 3, An overview of the study area. The reservoirs in the study area are labelled as A , B, C, and D, representing Harangi, Hemavathi, Kabini, and KRS reservoirs, respectively. The labels CA, CB, CC, and CD are used to denote the respective command areas associated with these reservoirs

Figure 5 is conceptual and therefore, it is merged with another figure as indicated below

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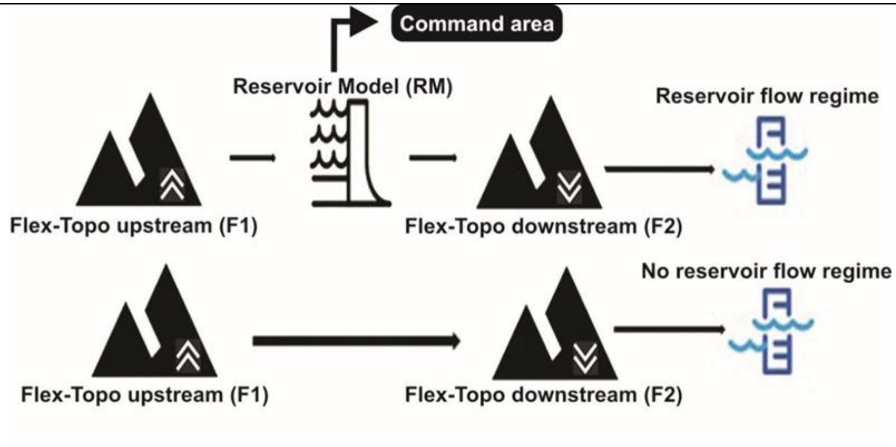
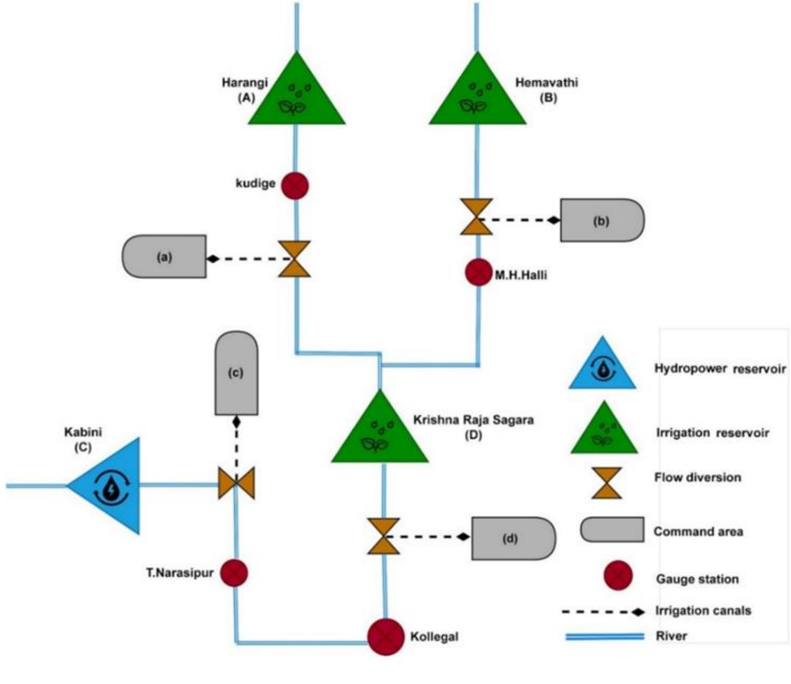


2 (a) Source : Ekka et al., 2022



2 (b)

	2(a) Modelling concept for individual reservoir: Upstream and downstream contributing areas of the gauging station (GS) are modelled as F1 and F2 respectively. The top row shows how the reservoir model (RM) that contributes to irrigating a certain Command area is integrated with F1 and F2 and calibrated. To simulate the pre-dam situation, RM is removed from the calibrated model, along with its contribution to irrigate the command area 2(b) Integration of reservoir in the basin: All the reservoirs are integrated together to assess the effect of reservoirs on the flow downstream for varying configurations of considered reservoirs. Various configurations of the reservoirs were considered to create a total of 16 different scenarios to assess the potential impacts of flow alterations on the river ecosystem services
In conclusion, in the opinion of this reviewer, a major revision of the methods and analysis is required for the publication of the paper. Likewise, supplementary materials with descriptions and data are required to illustrate with sufficient detail each of the analytical components developed. Supplementary data sets to allow for reproducibility are strongly encouraged.	A major revision of the paper has been made in response to the comments received. A supplementary dataset has been included and will be made available along with the manuscript.

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