Monitoring ecosystem services requires a redesign of siloed monitoring programmes

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Abstract

1. Monitoring ecosystem services is essential for achieving sustainability and biodiversity goals, yet existing monitoring programmes are fragmented, siloed, and not designed to detect or attribute change in ecosystem services.
2. We applied the Essential Ecosystem Service Variables (EESV) framework within a social-ecological network model to integrate three decades of ecological, economic, and social monitoring data from the Pacific salmon fisheries of British Columbia, Canada. Using Bayesian state-space models, we analysed the coupled provisioning (commercial) and cultural (recreational) services provided by five salmon species across six regions.
3. Our models revealed complex, species- and location-specific dynamics, including regional declines in Chum salmon abundance, long-term reductions in commercial

fishing effort, and diverging trends between commercial and recreational harvests, with recreational catchability consistently higher than commercial catchability.

4. Trade-offs between provisioning and cultural services were particularly evident for Chinook and Coho salmon, where recreational and commercial harvest rates displayed opposing trends, highlighting competition among user groups.
5. The modelling process exposed the limitations of current monitoring systems: many model structures failed to converge, key external drivers (e.g. sea surface temperature and hatchery releases) could not be reliably incorporated, and predictive accuracy was consistently poor for anthropogenic and governance components, demonstrating that existing monitoring programmes cannot support confident causal attribution of change.
6. Despite these limitations, the integration of siloed datasets recovered known dynamics and provided valuable insights, showing that social-ecological network models can serve both as analytical tools and diagnostics of monitoring capacity, providing an empirically supported mandate for the fundamental redesign of monitoring systems. To effectively manage ecosystem services and meet global sustainability targets, nations must move beyond fragmented data collection and build integrated, holistic monitoring programs that co-measure ecological, social, and governance variables by design, enabling an evidence-based understanding of our planet's vital human-nature systems.

Keywords

Monitoring, essential variables, social-ecological network, ecosystem services, Pacific salmon monitoring

Introduction

Healthy biodiversity and ecosystems provide an essential set of services for human wellbeing (Richardson *et al.*, 2023). Increasing human activity has led to a global decline in both biodiversity and ecosystem services (Brauman *et al.*, 2020), prompting a political response from nations around the globe. Specifically, the Global Biodiversity Framework (GBF) calls for a whole-of-society approach to transform humanity's relationship with nature (CBD, 2022). The GBF calls for nature's contributions to people, including ecosystem services, to be valued, maintained and enhanced, with those currently in decline to be restored. Delivering on this goal by 2050 requires managing ecosystem services, which depends on measuring the trends in their state and condition. The GBF includes a global monitoring framework that sets out how nations should measure change in biodiversity and ecosystem services (Affinito *et al.*, 2024). Yet, the challenge of effectively implementing monitoring is substantial as many gaps remain in the monitoring framework, especially for ecosystem services (Affinito, Butchart, *et al.*, 2025).

Effective ecosystem services monitoring requires recording both the ecological and societal components of ecosystem services through time. To date, the focus has been on ecosystem service assessments rather than monitoring (Seppelt *et al.*, 2011; Galaz García *et al.*, 2023). These assessments are typically single-time point estimates of ecosystem service supply, calculated from models using production functions that describe how biophysical inputs produce ecosystem services (Tallis and Polasky, 2009). While useful to understand where ecosystem services are produced and how this connects to people's demand, these assessments do not inform on the changing state of ecosystem services and are not the result of planned monitoring efforts to understand where, when and why ecosystem services are changing. Additionally, these assessments are typically done at very large scales (Chaplin-Kramer *et al.*, 2022), limiting their ability to inform local policy efforts (Mandle *et al.*, 2021), which is the scale at which ecosystem services are experienced by people.

Effective management of ecosystem services requires the implementation of monitoring programmes that are designed to detect change and attribute it to causal drivers at different scales (Gonzalez, Chase and O'Connor, 2023). Delivering on the GBF's ambition requires that monitoring go beyond simply tracking ecological state variables. Monitoring systems must be designed to understand the causes of change and opportunities for management from local to regional scales. In the context of ecosystem services, this requires monitoring the multiple dimensions of ecosystem services and how these are changing in relation to each other and to external drivers.

This can be achieved using the essential ecosystem service variables (EESV) framework (Balvanera *et al.*, 2022). The framework sets out which aspects of an ecosystem service need to be monitored to effectively understand change through six classes of variables: ecological supply, use, demand, anthropogenic contribution, instrumental value and relational value. The framework sets the conceptual foundation of how to monitor ecosystem services, but it remains to be systematically implemented in purpose-built monitoring systems. The EESV framework considers ecosystem services as part of social-ecological systems. That is, when ecosystem processes contribute to human activities (e.g. pollination and agriculture, water filtration and swimming), this realises an ecosystem service (Affinito, Holzer, *et al.*, 2025). Thus, monitoring of ecosystem services requires tracking variables in all relevant parts of the social-ecological system.

Conceptualising ecosystem services within social-ecological systems has proven effective for understanding human–nature interactions (Ostrom, 2007), predicting management outcomes

(McGinnis and Ostrom, 2014), and integrating diverse disciplinary data into a cohesive framework (Hinkel *et al.*, 2015). Building on this foundation, researchers have begun framing ecosystem services as spatially explicit social-ecological networks, where interacting social and ecological nodes drive system-wide dynamics (Dee *et al.*, 2017; Felipe-Lucia *et al.*, 2022). Though conceptually developed (Reyers *et al.*, 2013), empirical applications remain limited (Stanworth, Peh and Morris, 2024). A spatially explicit, multilayer network approach offers a way to monitor ecosystem services by identifying distinct layers (Figure 1) and tracking interactions across them (Leslie *et al.*, 2015). This structure captures the flow of ecosystem services and reveals how social and ecological dynamics interact across scales (Sun *et al.*, 2019), enabling the identification of scale mismatches and supporting causal inference (Bodin *et al.*, 2019). This approach provides the blueprint needed to integrate existing disparate data sources into a cohesive understanding of ecosystem service change (Firkowski *et al.*, 2021).

Existing monitoring programmes already collect vast amounts of data that are relevant to ecosystem services (Tallis *et al.*, 2012). The EESV framework and social-ecological network theory provide the conceptual framing necessary to take advantage of these data and test their appropriateness for ecosystem service monitoring. The EESV framework helps to identify which variables to focus on and social-ecological network theory indicates how to relate these variables to analyse their dynamics. This provides an opportunity to test whether existing monitoring programmes can be linked together using EESVs and social-ecological network theory to monitor ecosystem service dynamics.

We implement a social-ecological network analysis of the data-rich wild Pacific salmon fisheries from British-Columbia (BC, Canada) where Affinito, Fortin and Gonzalez (2025) already identified EESVs for the region. In BC, economic, social and environmental monitoring programmes with distinct mandates each collect data relevant to the services provided by wild Pacific salmon. These distinct mandates lead to monitoring choices that produce disjointed observations not designed *ad hoc* to be appropriate for social-ecological network analysis. We specifically test if an integrated network model can reveal location- and species-specific differences in the dynamics of four dimensions (ecological supply, use, anthropogenic contribution, and value) of two coupled ecosystem services (commercial provisioning and recreational cultural fisheries). We combine data from multiple monitoring programmes to conduct an analysis of the social-ecological network using mechanistic models fitted using a Bayesian approach. First, we test whether by connecting these monitoring systems into one social-ecological network we can recover expected mechanistic relationships to identify key ecosystem service dynamics. Second, we discuss the strengths and weaknesses of current monitoring efforts, and our ability to monitor ecosystem services by combining datasets collected from siloed monitoring programmes.

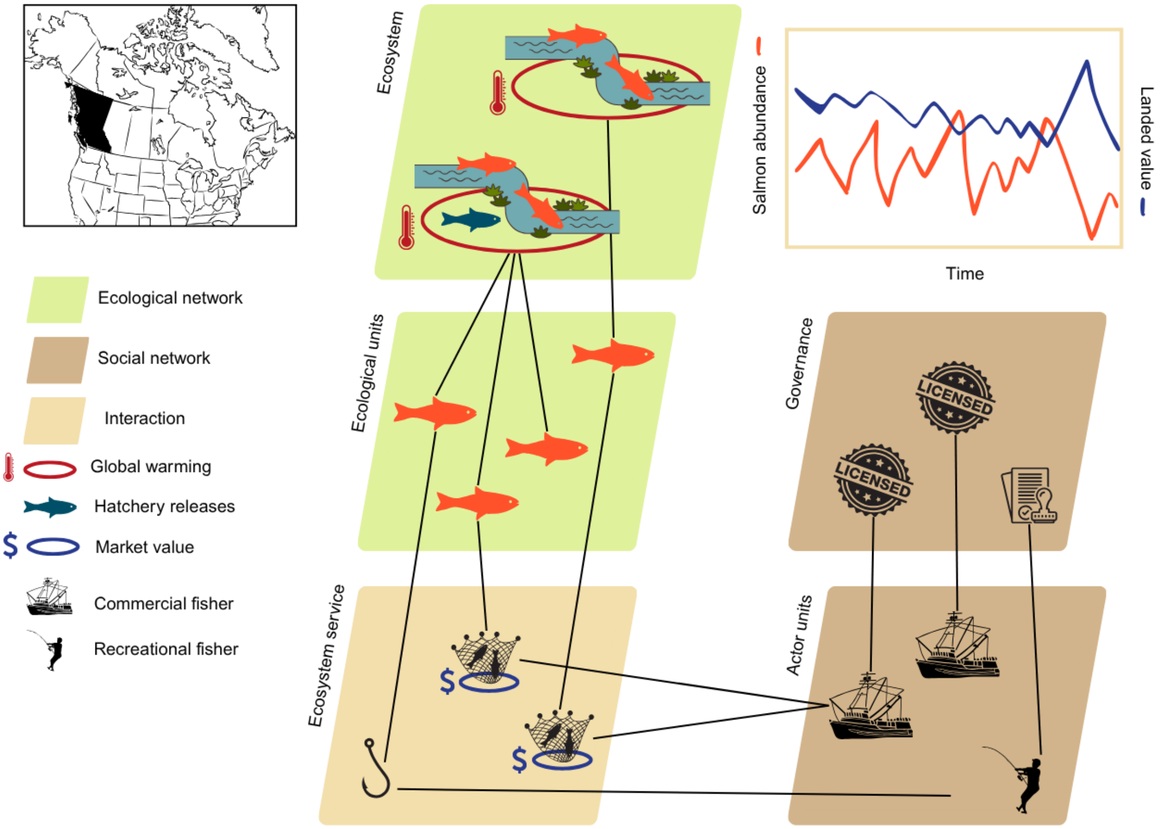

Figure 1. The social-ecological network of Pacific salmon fisheries in British-Columbia provides multiple ecosystem services (both cultural and provisioning). Social and ecological nodes interact to produce ecosystem services. These interactions are mediated by external drivers (e.g. warming and hatchery releases) and governance rules (e.g. licensing). Each of these nodes, drivers and interactions can be monitored to understand how different dimensions of the ecosystem service are changing. Figure inspired and adapted from Firkowski *et al.*, (2021).

Methods

Study system

Coastal waters of British Columbia, Canada, host one of the world's largest animal migrations each year (Groot and Margolis, 1991). Five species of Pacific salmon – *Oncorhynchus tshawytscha* (Chinook), *O. kisutch* (Coho), *O. keta* (Chum), *O. gorbuscha* (Pink), and *O. nerka* (Sockeye) – return from the ocean to lay eggs in streams along the coast.

Three major fisheries rely on Pacific salmon: Indigenous, recreational, and commercial. Catch quotas are allocated to ensure that enough fish return to spawn, giving Indigenous and recreational fisheries priority access. The commercial salmon industry is a source of income and identity for many and recreational fisheries attract anglers from around the world. Together, these fisheries generate provisioning (commercial) and cultural (recreational) ecosystem services that are central to the province's social fabric and ecological integrity. Canada's Wild Salmon Policy (DFO, 2018) aims to safeguard wild stocks through a mix of conservation measures and stock enhancement via hatcheries. The result is a complex social-ecological network (Figure 1), in which multiple services are co-produced.

Multiple organizations collect and report monitoring data. The federal government monitors licenses, catch, and effort in commercial and recreational fisheries. The provincial government records landings and revenue. Indigenous catch data remains sparse. The Pacific Salmon Foundation, a local NGO, processes abundance estimates and hatchery release data. Remote sensing programmes, like the EU's Copernicus services, monitor ocean conditions. These various monitoring efforts differ in scale, timing, and spatial resolution, and are not coordinated across and within organizations. Specifically, salmon abundance is monitored in spawning streams to estimate numbers. These data are then processed by the Pacific Salmon

Foundation who reports abundance and hatchery release estimates for each salmon population into six regions corresponding to major river systems in BC (Haida Gwaii (HG), Nass, Skeena, Central Coast (CC), Vancouver Island and Mainland Inlets (VIMI) and Fraser). Effort, license and catch data are collected by the federal Department of Fisheries and Oceans (DFO) at sea according to a set of geographically defined management areas. Landing and resale value data are collected by the provincial government and reported at the provincial scale. Remotely sensed data are available from Copernicus daily at high resolution.

Data acquisition and processing

We acquired the data from each organisation (Supplementary Table S1). These data correspond to four essential ecosystem service variables (EESVs) for commercial fisheries (ecological supply, use, anthropogenic contribution and instrumental value) and three EESVs for recreational fisheries (ecological supply, use and anthropogenic contribution) as well as three external drivers (sea surface temperature, hatchery releases, and licensing). We limited our analysis to the years 1996-2023 as most datasets included no data earlier than 1996. Even within the study period, data gaps in several time series were present due to changes in funding for local monitoring or responsibility for compiling monitoring data and making it available.

Due to the mismatch in monitoring scale and location, we had to process the data to spatially match the variables to each other. We followed the method used by the Pacific Salmon Foundation to match catch to abundance counts (Pacific Salmon Foundation, 2024). That is, we (i) mapped DFO management areas to each river system in the Pacific Salmon Foundation's dataset based on salmon migration routes, (ii) calculated the proportion of spawners for each species in each region that corresponds to each management area in any given year and (iii) multiplied the catch of each species in each management area on any given year by the proportion of spawners in each river system to assign catch numbers measured in a management area to its corresponding river system. This methodology assumes that fish from different river systems are caught in proportion to their abundance. Similarly, we assigned landed value to each river system by calculating the proportion of catch of each species from each region and multiplying it by the total provincial scale landed value of each species.

We matched effort and licenses to each river system by assigning total effort and total license numbers to their corresponding region. We did not use a proportional approach for effort, fleet size and license counts as these are not species-specific but rather apply to all species equally. For sea surface temperature, we used the average temperature of coastal BC waters in the months that smolts migrate out to sea.

Analytical approach

We used a set of Bayesian state-space models to analyse the disjointed monitoring data from the social-ecological network. State-space models make modelling complex social-ecological network dynamics possible by separating the underlying system dynamics from the data measurement process (De Valpine and Hastings, 2002). Key variables in the system are modelled as unobserved latent states. These latent states represent the true unobserved state(s) of the system (e.g. the number of salmon returned to spawn before harvest) and link to the data through the observation model, which draws from a distribution centered on the latent states plus some observation error. Furthermore, this approach handles missing data. For years where observations are absent, the model still estimates a latent state, which is informed by the process model's dynamics and data available from other years and variables. Finally, the model allows for partial pooling, where parameters for each individual location are assumed not to be independent. This allows the model to "learn" parameter values for each location from other locations, enabling information sharing and estimation of system dynamics even for regions where monitoring effort was lower.

We defined a set of 15 mechanistic state-space models of increasing complexity to model the social-ecological network (Table 1). At its core, each model reflects the relationships between salmon abundance and commercial catch and fishing effort. To this, we added each additional variable with its corresponding mechanistic process linking it to others. All models were written in Stan version 2.36.0 (Carpenter *et al.*, 2017) and fit to each of the five salmon species independently.

Table 1. State-space models tested in the Pacific salmon social-ecological network with variables included in the model. Each of these models was fit to each species.

Model name	**Variables included**
CSE	Commercial catch & effort + Spawner abundance (CSE)
CSEF	CSE + Commercial fleet size (CSEF)
CSEFV	CSEF + Landed value of commercial catch (CSEFV)
CSEFVH	CSEFV + Hatchery releases
CSEFVHTemp	CSEFV + Hatchery releases + Sea surface temperature
CSEFVR	CSEFV + Recreational catch & effort
CSEFVRTemp	CSEFV + Recreational catch & effort + Sea surface temperature
CSEFVRStp	CSEFV + Recreational catch & effort + Recreational licenses
CSEFVL	CSEFV + Commercial licenses

CSEFVLH	CSEFV + Commercial licenses + Hatchery releases
CSEFVLHTemp	CSEFV + Commercial licenses + Hatchery releases + Sea surface temperature
CSEFVLR	CSEFV + Commercial licenses + Recreational catch & effort
CSEFVLRTemp	CSEFV + Commercial licenses + Recreational catch & effort + Sea surface temperature
CSEFVLRStp	CSEFV + Commercial licenses + Recreational catch & effort + Recreational licenses
CSEFVLRStpH	CSEFV + Commercial licenses + Recreational catch & effort + Recreational licenses + Hatchery releases

Model specifications

The process model used reflects well-established mechanistic relationships between key variables in the social-ecological network of Pacific salmon. The Ricker equation (Ricker, 1954) models the spawner-recruit relationship of salmon:

$$R = \alpha S e^{-\beta S}, \tag{1}$$

where R is the number of recruits produced by the spawners S with a maximum number of expected spawners per recruit (*i.e.* productivity parameter) α and density-dependent effect β. The Ricker model can be extended to accommodate the effects of hatchery releases and sea surface temperature to:

$$R = \alpha S e^{-\beta S + \gamma H + \delta T}, \tag{2}$$

where H is hatchery release and T is sea surface temperature and γ and δ are their respective effects contributing to recruitment or reducing survival. Using the Schaefer model (Schnute,

1977) and Baranov's catch equation (Quinn and Deriso, 1999), recruitment and spawners can be linked to catch via harvest:

$$S = (1 - h)R \, , \tag{3}$$
$$C = hR \, , \tag{4}$$

where C is catch and h is the harvest rate parameter. Harvest rate is a saturating function of effort with:

$$h = \frac{qE}{1 + qE} \, , \tag{5}$$

where E is effort and q is catchability (*i.e.* how easy it is to catch salmon for any given unit of effort). These models can be extended to include recreational catch through the same equations as (4) and (5) but for recreational catch and connected to R and S through:

$$S = R \, (1 - \; h_c - h_r) \, , \tag{6}$$

where h_c and h_r are commercial and recreational harvest rates, respectively. Fleet size affects effort through a power law function:

$$E_c = \; \phi_1 F^{\phi_2} \, , \tag{7}$$

where E_c is commercial effort, F is fleet size and ϕ_1 is the effort scaling parameter determining the overall magnitude of effort on fleet size and ϕ_2 is the fleet-effort elasticity establishing whether there are diminishing returns (<1) or intensification (>1). Commercial licensing controls fleet size with:

$$F = a_c L_c \, , \tag{8}$$

where L_c is the number of licenses and a_c is the proportion of licenses that are actively being used. Recreational licenses, however, control effort directly with:

$$E_r = a_r e_r L_r \, , \tag{9}$$

where E_r is recreational effort, L_r is the number of recreational licenses, a_r is the participation rate and e_r is the average effort per active recreational license. Finally, the economic value of commercially landed salmon follows a bioeconomic allometric relationship with catch:

$$V = \upsilon C^{\omega} \, , \tag{10}$$

where V is value, υ is the unit value coefficient reflecting the value per unit catch and ω is the value elasticity.

These equations can be linearized and reparametrized in log space to allow for model fitting using an MCMC sampler. All parameters vary per species through space (*i.e.* across all six regions, except for Chum salmon where no data on catch nor abundance were available for Fraser) and some through time (h_r and h_c). See Supplementary Tables S2 & S3 for complete model specification.

Model fitting

We fit all fifteen models to each species using the *cmdstanr* package version 0.9.0 (Gabry *et al.*, 2025) in R version 4.3.1 (R Core Team, 2022). Each model was run for a total of 20,000 iterations with 10,000 warmups. We assessed model convergence via visual inspection of chains and $\hat{R}$ statistics (Brooks and Gelman, 1998). Any model or parameter where chains were not well mixed and/or $\hat{R}$ values were too high (>1.1) was deemed to not have converged. We further checked effective sample sizes for all converged models to check for adequate sampling of the posterior (Raftery, Lewis, and others, 1992). To reduce problems linked to identifiability, multimodality and general non-convergence we used informative priors where justifiable, removed redundant explanatory variables, used a non-centered approach for all hierarchical parameters and centered several predictor variables.

We compared converged models using visual inspection of posterior predictive plots. We further computed a Bayesian alternative to R^2 (Gelman *et al.*, 2019) for each response variable in the model. This is a model-based statistic that calculates the proportion of variance in future data explained by the model. We considered a model better than its alternative when convergence parameters were good, its posterior predictive plots overlapped better with the data and its Bayesian R^2 values were higher. We further assessed the predictive ability of our best-fit models for each species using leave-one-out cross-validation, which estimates how well the model predicts each observation when it is omitted from the fitting process (Vehtari, Gelman and Gabry, 2017).

Results

Our analysis of BC's Pacific salmon fisheries produced two complementary sets of findings. First, by applying a social-ecological network model to disparate monitoring datasets, we successfully uncovered complex dynamics, including species- and region-specific abundance trends and opposing dynamics between cultural and provisioning services. Second, the modelling process itself served as a diagnostic tool, with model convergence failures and poor predictive accuracy revealing the fundamental insufficiency of the current, siloed monitoring programmes to support monitoring of the full social-ecological dimensions of these ecosystem services.

Social-ecological network dynamics

Species-specific dynamics in ecosystem service trends

We focus on and report the findings from the best-fitting model for each species (Table 2; Supplementary Table S4). Parameter and uncertainty estimates are available in Supplementary Table S6.

The best-fit model for Sockeye salmon was the simplest, including only fleet size, effort, catch, and abundance. Regional abundance estimates varied, with CC, Nass, Skeena, VIMI showing similar annual estimates, HG showing lower estimates, and Fraser showing markedly higher and more variable estimates, characterized by pronounced boom–bust cycles (Supplementary Figure S1a). Commercial catch declined across all regions, although estimates were highly uncertain, especially in Fraser, where catch trends did not clearly reflect abundance fluctuations (Supplementary Figure S1b). Despite this, Fraser had the highest estimated catch, followed by Nass, Skeena, and VIMI, with the lowest estimates in CC and HG. Catchability and harvest rates were the lowest in CC and HG, moderate in Fraser and VIMI, and highest in Nass, although credible intervals for catchability overlapped across several regions (e.g. Nass and Skeena; Skeena, VIMI, and Fraser; Fraser and CC).

Chum salmon was the only species for which the best-fit model supported a decline in abundance in two regions (HG and Skeena; Figure 2a), driven by negative values in the productivity parameter (α), though credible intervals overlapped zero in both cases (Figure 2b). CC had a positive α value, but its credible interval also included zero. Abundance estimates were highest in CC and VIMI, followed by HG, Nass and Skeena. Catch estimates declined across all regions, with wide credible intervals, especially in HG and Skeena (Supplementary Figure S2b). This decline in catch was accompanied by a corresponding decline in landed value

(Supplementary Figure S2c). Value elasticity was consistently less than 1 across regions (CC: 0.71; HG: 0.72; Nass: 0.68; Skeena: 0.87; VIMI: 0.74), suggesting diminishing returns of catch on income, though intervals overlapped with 1 for Skeena.

Abundance estimates for Chinook were highest in Fraser and VIMI, moderate in Skeena and CC, and lowest in Nass and HG. Uncertainty was the greatest in CC and Skeena, despite data gaps in HG post-2007 (Supplementary Figure S3a). Catch estimates were broadly similar across regions, except for a decline in CC after 2018 (Supplementary Figure S3b). Landed value mirrored these trends, with a sharp decline in CC since 2018 and a consistent dip across all regions (except Fraser) in the late 1990s and early 2000s (Supplementary Figure S3c). Value elasticity estimates were below 1 across all regions, although their credible intervals included 1. Effort elasticity was below 1 for all regions (CC: 0.84; Fraser: 0.84; HG: 0.88; Nass: 0.86; Skeena: 0.86; VIMI: 0.84), indicating diminishing returns in effort with increasing fleet size.

The Coho model revealed similar trends across abundance, catch, and value (Supplementary Figure S4). Regions with higher abundance (CC, Skeena, Nass and HG) also had higher catch and value. Fraser and VIMI had the lowest estimates across all three metrics. While abundance appeared to increase across regions over time, this trend was not reflected in catch or value in CC, Nass, Skeena, or Fraser, where post-2016 declines were observed. Value elasticity was below 1 in all regions, with credible intervals overlapping 1.

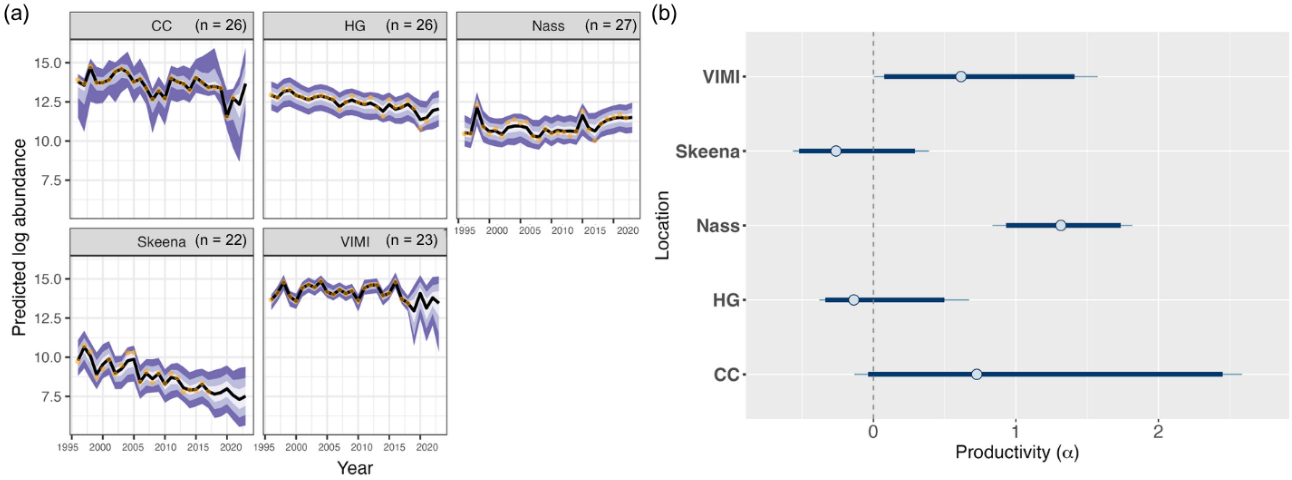

Figure 2. An uncertain regional decline in Chum salmon abundance. (a) Predicted decline in Chum abundance in the Skeena and HG regions is related to (b) a negative productivity parameter (α) leading to a growth rate smaller than 1. Note the overlap with 0 of credible intervals of the α parameter in both regions. All other regions do not show a decline, except that CC's α parameter credible intervals also overlap 0.

Competing cultural and provisioning services

We found a decline in commercial fishing effort across all species and regions, particularly in the North (Nass, Skeena, HG, and CC), where the decline began in 1996 (Figure 3a). Recent rebounds in effort were observed only in Nass and Skeena, with continued decline in HG and CC. In contrast, the South (Fraser and VIMI) did not exhibit a notable decline until 2019, after which effort dropped and did not recover. Across all species, commercial effort estimates showed the lowest uncertainty of any modelled variable and did not differ substantially between regions. For Chinook and Coho, where recreational data were included, no equivalent decline in recreational effort was found (Figure 3b). Recreational effort also appeared evenly distributed across regions. However, the models struggled to fit recreational effort in Fraser and VIMI, resulting in high uncertainty in those regions.

In contrast, recreational catch estimates did not exhibit high uncertainty in Fraser or VIMI (Supplementary Figure S5). Uncertainty was only pronounced for Chinook recreational catch in CC. Recreational catch was generally stable from 2005 onward, except for a notable dip in 2020 for Chinook and Coho in HG, and for Coho in CC. Prior to 2005, a steady increase in recreational catch was observed, though this was likely a modelling artefact due to missing data and periods of lower estimated abundance. Notably, commercial and recreational harvest rates exhibited opposing trends, suggesting competitive dynamics between these sectors (Figure 3c). Additionally, recreational catchability exceeded commercial catchability across all regions for both Chinook and Coho, with non-overlapping credible intervals (Figure 3d).

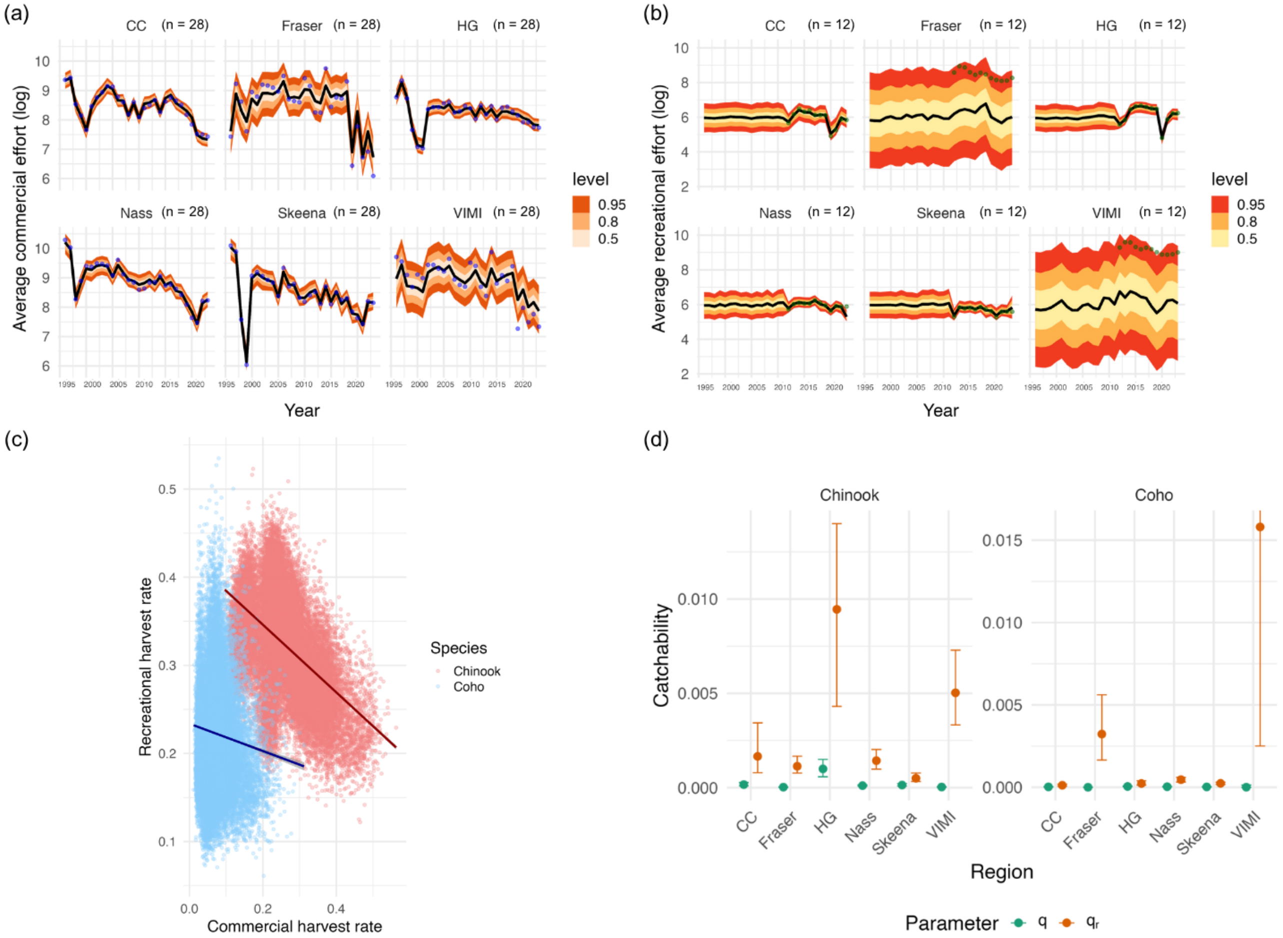

Figure 3. Contrasting dynamics of commercial and recreational fishing for Chinook and Coho salmon. (a) Average commercial fishing effort for both species in each of the six regions shows a long-term decline in the North and a recent decline in the South. (b) Average recreational fishing effort for both species shows no trend over time, with wide uncertainty in the South. (c) Recreational and commercial harvest rates for both Chinook and Coho show a negative relationship (1000 parameter draws shown with linear model line fitted, uncertainty bounds around the line are shown in grey). (d) Recreational catchability is consistently higher than commercial catchability for both species in most regions. The Coho plot is cut at 0.016 due to large uncertainty in VIMI estimate (up to $q_r = 0.165$).

Model performance as a diagnostic

Data integration and model stability

The limitations of the existing monitoring data were apparent as most of the fifteen candidate model structures failed to converge for most species (Table 2). Notably, no models converged for Pink salmon. Simpler structures worked best for Chum and Sockeye. Chinook and Coho required a more complex model that included recreational catch and effort. None of the models that included sea surface temperature as a predictor converged for any species. Certain covariates consistently degraded model performance: models with hatchery releases or commercial license data often failed to converge or, when they did, produced poor fits and wide, uninformative uncertainty bounds. Further complexity was detrimental; adding recreational licenses or combining them with commercial license data led to model failure or poor performance.

Poor predictive power for social dimensions

Cross-validation of the best-fit models revealed consistent differences in predictive accuracy across variable types. Models performed well for commercial catch and landed value, and moderately well for spawner abundance, though performance varied by species (Supplementary Table S5). In contrast, predictive accuracy was consistently poor for anthropogenic components (*i.e.* commercial effort, fleet size, and licensing) and recreational components across all species. For example, in Chinook salmon, while over 80% of observations for spawner abundance and economic metrics had acceptable *k*-values ($k<0.7$), more than 77% of the values for effort and recreational metrics were poorly predicted ($k>0.7$). This pattern held across species.

For Chum and Coho salmon, models showed good predictive accuracy for commercial catch and value, but poor performance for effort, fleet size, and recreational components, where the majority of *k*-values exceeded 0.7. The Sockeye model's predictive ability predicted commercial catch with high accuracy (98.8% *k*<0.5) but failed to do so for effort and spawner abundance (more than 68% of the values were poorly predicted (*k*>0.7)). Overall, the models effectively captured biological and economic components of the system but showed weaknesses in modeling anthropogenic contribution and governance.

Table 2. Model convergence and fit for all salmon species. Models are ordered by complexity. Dashes indicate failure to converge. Ranking describes model fit from best to worst.

Model	**Species**				
	Chinook	Chum	Coho	Pink	Sockeye
CSE	-	2nd	-	-	-
CSEF	2nd	3rd	-	-	1st
CSEFV	3rd	1st	2nd	-	2nd
CSEFVH	-	-	-	-	5th
CSEFVHTemp	-	-	-	-	-
CSEFVR	1st	-	1st	-	4th
CSEFVRTemp	-	-	-	-	-
CSEFVRStp	-	-	-	-	3rd
CSEFVL	6th	-	-	-	-
CSEFVLH	-	-	-	-	-
CSEFVLHTemp	-	-	-	-	-
CSEFVLR	5th	-	-	-	-
CSEFVLRTemp	-	-	-	-	-

CSEFVLRStp	4th	4th	-	-	6th
CSEFVLRStpH	-	-	-	-	-

Discussion

This study provides the first demonstration that siloed monitoring data can be integrated through a social-ecological network approach to assess ecosystem service dynamics. Despite inherent limitations in drawing from disparate monitoring systems, the models uncovered nuanced, species-specific dynamics. By applying the EESV framework to BC Pacific salmon fisheries, we recovered patterns in abundance, catch, and value across species and regions, and identified trade-offs between provisioning and cultural services. Critically, the modelling process served not only as an analytical tool but also as a diagnostic, revealing that current monitoring infrastructure is not yet capable of supporting integrated, mechanistic understanding of change.

Ecosystem services dynamics in a coupled system

Our analysis of EESVs revealed pronounced species- and region-specific dynamics. For example, Chinook and Sockeye showed higher abundance, catch, and value in the South, while they were higher in the North for Coho, a pattern also reflected in recreational catch. These differences highlight how each species contributes distinctly to provisioning and cultural ecosystem services in the province, consistent with the portfolio effect established in Pacific salmon (Schindler *et al.*, 2010). This effect, shaped by species and location diversity (Griffiths *et al.*, 2014), underpins the sustainability of salmon-derived services (Moore, Connors and Hodgson, 2021) and suggests that they are not uniform but should be understood as a bundle of services with varying dynamics (Raudsepp-Hearne, Peterson and Bennett, 2010). The need for different model structures across species further supports this, underscoring that effective management must be tailored to the unique social-ecological context of each species. While current policy treats spawning stocks as distinct conservation units (DFO, 2018; Grant, MacDonald and Winston, 2019), the continued use of mixed-stock fisheries creates a disconnect between conservation and resource management that must be addressed.

A consistent pattern across species was the decline in commercial fishing effort, particularly in Northern regions. In contrast, the decline was less pronounced in the South, where most of BC's population resides and where multiple conservation units, especially for Chinook and Sockeye, are in decline or at risk (Price *et al.*, 2017). The more recent drop in effort in the South may reflect increased urgency to act through temporary fishery closures. Notably, there was no similar decline in recreational effort, and recreational catch levels remained comparable to commercial catch across regions. This highlights a disconnect in current management, which has focused on commercial fisheries while overlooking recreational pressures. Making these opposing pressures visible is a direct result of the social-ecological network approach, which models the system as a whole rather than assessing each fishery in isolation.

Our analysis also revealed apparent competition between the provisioning and cultural ecosystem services derived from Pacific salmon. While both fisheries draw on the same ecological supply, they differ in beneficiaries, use, and anthropogenic contribution, resulting in opposing harvest rates, particularly for Chinook, a highly prized target for anglers known as "king salmon". These dynamics align with federal policy that prioritizes access first to Indigenous Peoples, then recreational, and finally commercial fishers. This highlights the importance of analysing connected ecosystem services together, as single-service assessments risk missing such interactions (Renard, Rhemtulla and Bennett, 2015; Meacham *et al.*, 2022).

Effective monitoring must therefore account for the interplay among services to inform sustainable management.

The anatomy of a siloed system: lessons from the data

A significant barrier to the advancement of monitoring ecosystem services is the challenge of integrating ecological, social and economic data (Bennett *et al.*, 2015). These disciplines have historically worked independently, focused on each element of the system in isolation. Our analysis of social and ecological systems as interdependent and tightly coupled, supported by EESVs, has the potential to tackle this challenge even for cases where the data is not intended for that purpose.

Using EESVs and some defensible assumptions, we were able to handle the spatial mismatch in data collection efforts to conduct our analysis. For example, we assumed that catch numbers would be spread evenly between spawning stocks although some stocks may be more targeted than others due to run timing, random chance or population-specific migration strategies (Byron and Burke, 2014). Similarly, we assigned value, to the respective proportion of catch each year, assuming that there are no differences between regions that could lead to the price per fish changing. This is unlikely, as port proximity and ease of access likely affect the price of fish in more remote regions. Additionally, no age structure of spawners is available in Canadian datasets, although it is important information for mechanistic models of salmon (Fleischman *et al.*, 2013). Finally, we used an average for sea surface temperature when spawning river temperatures or more specific locations for sea surface temperature would likely be more relevant. Such data are accessible from remote sensing organisations, but no effort has been made to systematically relate the information on temperature to spawner counts. While violation of some of these assumptions may be linked to convergence issues (e.g. temperature) or poor model fit (e.g. value), several models converged, fit, and had good explanatory power, suggesting that they were appropriate in representing the network for those species. Thus, even with imperfect data, integrated social-ecological network modelling can yield valuable insights and highlight where future improvements are needed to target the most limiting data gaps.

The large uncertainty estimates from our models are not a methodological limitation, but rather a direct and quantifiable reflection of known data quality issues within the monitoring programmes themselves. These challenges include historical funding cuts leading to data gaps for specific species and locations, and the fact that the recreational catch program (Internet Recreational Effort and Catch – iREC) only began in 2012. Furthermore, monitoring programmes themselves make simplifying assumptions to report estimates. iREC extrapolates estimates from voluntary self-reported data with historically low completion rates (30-50%). While our state-space model was chosen specifically to handle such issues, its higher uncertainty in those years is an honest reflection of decreased confidence, a finding consistent with external reports of declining data reliability since 2014 (Atkinson *et al.*, 2024). Therefore, rather than simply limiting management potential, this quantified uncertainty is essential information. It provides an evidence-based case for where monitoring investments are most critically needed and highlights the specific ecosystem components where management decisions are currently being made with the greatest risk.

A diagnostic tool for monitoring systems

The performance of the models served as a diagnostic of the limitations of the monitoring programme. Widespread issues with model convergence and fit were not random failures as they consistently occurred when the analysis incorporated key external drivers or specific governance mechanisms. The importance of external drivers in ecosystem service dynamics is well established (Dade *et al.*, 2019). Yet, those models that did include such drivers failed to converge, suggesting that the current monitoring data is insufficient to confidently attribute changes in salmon populations to their purported drivers in the network. Moreover, no models converged for Pink salmon. Pink salmon are the only species with a fixed two-year spawning cycle and are therefore typically modelled as even or odd populations (Pacific Salmon Foundation, 2024). Our models do reflect this cycle, but it is possible that modelling the species as a single ecosystem service, mixing data for both odd and even populations, fundamentally misspecifies the real relationships, suggesting that Pink salmon management should not treat this species in a monolithic manner but rather as it would two different species.

Additionally, the high uncertainty in many parameter estimates, even in converged models, is a direct consequence of monitoring programmes not designed for this type of integrated analysis. We were unable to confidently state that some Chum salmon populations are in decline. However, work done by the Pacific Salmon Foundation reveals that all eight Chum conservation units in the Skeena and HG regions are in long-term decline (Pacific Salmon Foundation, 2024). Moreover, they report that the data on these populations is of medium to low quality. This corroborates our findings and indicates that the model did recover the correct trends. Additionally, the model including recreational catch fit best for Coho and Chinook salmon. These species are prized fish for anglers who tend to focus their efforts on them. Our social-ecological network models recovered this fact by excluding recreational catch for Chum and Sockeye. Thus, monitoring ecosystem services in social-ecological networks holds promise in supporting the detection and attribution of change.

Another key diagnostic is the ineffectiveness of our approach at accounting for governance rules. Including license controls resulted in a poorer fit for all models. This suggests our mechanistic assumptions about how fishers respond to regulations do not capture the complex socio-economic realities that drive participation in the fishery. Better connecting governance to ecosystem service dynamics is important for management (Barfuss *et al.*, 2018), especially if licensing is used as the primary tool to control the fisheries. More complex alternatives include the use of decision-based models to set harvest strategies (Barfuss *et al.*, 2017) or additional rules to control fishing effort. These models have the potential to represent socio-cultural dynamics and their effects on different dimensions of ecosystem services (Metzger *et al.*, 2021). This is particularly relevant in the context of interacting ecosystem services where different actors benefit or not as they compete, which we found to be the case for Pacific salmon. Thus, monitoring programmes aimed at supporting management must more effectively measure and account for the role of human behaviour in the social-ecological network.

Rethinking ecosystem services monitoring

If monitoring systems are to enable the detection and attribution of change, it is crucial that resources be redirected to transform siloed monitoring programmes into holistic monitoring systems that can operate across scales and disciplines (Gonzalez *et al.*, 2023; Vári, Gonzalez and Bennett, 2025). Our results show that modelling social-ecological networks using EESVs provides the conceptual framing for this in the case of ecosystem services, but questions remain.

Including relational values in ecosystem service monitoring is a particular challenge, as these are rarely measured (Schulz and Martin-Ortega, 2018). We did not include Indigenous people in the social-ecological network, although they are a key contributor to the sustainable management of Pacific salmon (Atlas *et al.*, 2021). Indeed, BC First Nations are already actively involved in monitoring salmon (Steel *et al.*, 2021) and efforts must be made to include them in the redesign of connected monitoring systems for ecosystem services. Additionally, it is possible that species of lesser economic value are being more poorly monitored as they do not reflect the information needs of the industry (Atkinson *et al.*, 2024), pointing out blind spots in the ability to manage ecosystem services through a narrow focus on economic valuation, to the detriment of other value systems (Pascual *et al.*, 2023). Alternative value systems are essential to recognise and consider, especially in the case of salmon where other cultural services embodied by important relational values play a central role for some actors (Himes *et al.*, 2024).

Moreover, ecosystem service monitoring systems must recognise that social-ecological networks are open systems. Here, we defined the boundaries of our system within BC. However, salmon are a highly migratory species and many populations swim through US waters, contributing to ecosystem services on the other side of the border and affecting ecological supply estimates. Salmon are also part of complex food webs and ecosystem-based modelling has proven effective in understanding fisheries (Geary *et al.*, 2020). Additionally, international markets can have telecoupling effects (Liu, Yang and Li, 2016) on salmon prices and drive dynamics of ecosystem service value in a way not captured by the current model. Using EESVs whilst considering the role of key internal and external drivers of the system (Schwantes *et al.*, 2024, 2025), would helps focus resources on those key variables to monitor within the social-ecological network.

Therefore, our findings serve as a powerful argument for the redesign of monitoring systems for ecosystem services. An effective system must be built on the principles of social-ecological networks, where social, economic, and ecological variables are monitored in a coordinated manner across relevant scales by and for all actors. Taking this approach requires fundamentally rethinking the role and function of monitoring systems. It means moving beyond insular agency mandates towards an integrated system designed to understand the links between ecosystem components and human activities, relying on monitoring standards such as EESVs. In practice, this would involve co-locating social and ecological data collection in space and time and ensuring data streams are designed from the outset to be interoperable. Adopting such a framework would allow for the confident attribution of change to specific drivers, empowering management efforts.

Conclusion

This study demonstrates both the promise and the remaining needs and challenges of monitoring ecosystem services. By applying a social-ecological network framework with EESVs, we successfully uncovered nuanced, interacting dynamics between species and fishery sectors that would be invisible to traditional, siloed approaches. However, our analysis also serves as a critical diagnostic, revealing that the fragmented and uncoordinated monitoring systems currently in place, even for highly monitored and valued ecosystem services in developed countries, are fundamentally ill-equipped to handle the complexity of ecosystem services. Without a deliberate and funded redesign of monitoring to explicitly capture the interconnected nature of people and nature, our ability to deliver on the ambitious goals of the Global Biodiversity Framework will be severely compromised, leaving the future of vital ecosystem services, like those provided by wild Pacific salmon, to chance.

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Data availability statement

Data and scripts for this article are available at https://github.com/FlavAff/SalmonSEN

Conflict of interest statement

The authors declare no conflict of interest.

CRediT authorship contribution statement

F. Affinito: Writing – review & editing, Writing – original draft, Visualization, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. M.-J. Fortin: Writing – review & editing, Supervision, Funding acquisition. A. Gonzalez: Writing – review & editing, Supervision, Resources, Funding acquisition, Conceptualization.

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