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RESEARCH

Updating Swedish hunting harvest estimates of open season game based on new methods and documented data

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Abstract

Reliable hunting bag statistics are central for informed wildlife management. In the absence of complete reporting, hunting harvest must be estimated based on partial data, which requires reliable data and appropriate statistical methods. In the Swedish system, hunting teams, whose positions are known to the level of Hunting Management Precincts (HMPs), report their harvest of open season game and the size of the land on which they hunt, and the harvest on the non-reported area is estimated based on the reports. In this study, we improved data quality by solving several identified issues in the spatial data and provided temporally consistent estimates of huntable land (EHL) based on documented assumptions. We applied a recently developed method, the Bayesian Hierarchical and Autoregressive Estimation of Hunting Harvest (BaHAREHH), to harvest reports of 34 species from 2003–2021, using both previous and updated EHL, and compared harvest estimates to previously available estimates using naïve linear extrapolation (LE), which has been used as Sweden's official harvest statistics. We found that updating EHL had a minor effect on harvest estimates at the national level but sometimes had a large impact at the level of individual HMPs. At the national level, previous LE estimates were similar to updated BaHAREHH estimates for species harvested at large numbers, but discrepancies were observed for species harvested at low rates. Time series of harvest estimated with LE had exaggerated temporal trends, higher coefficient of variation, and lower autocorrelation. At the level of counties and HMPs, there were substantial differences for all species, with some harvest estimates differing by several orders of magnitude. We conclude that the previously available LE estimates are sensitive to individual reports that add variability to the estimates and are, for some species, unreliable, especially at the level of county and HMP.

Keywords Hunting harvest · Harvest statistics · Hunting bag · Hierarchical Bayesian Modelling · Voluntary reporting

Introduction

Wildlife management depends on reliable estimates of game harvest (Elmberg et al. 2006; Aebischer 2019). Though harvest statistics are imperfect proxies for population abundance (Kahlert et al. 2015), they are often the only long-term data available (Smith et al. 2005; Aebischer 2019; Cretois et al. 2020). Harvest statistics are also an integral part of population models and projections (Rutten et al. 2019; Andrén and Liberg 2023).

Data availability and reporting systems vary with country and species. For instance, harvest reporting is entirely voluntary in the UK, Greece, and Cypress, voluntary for some species and mandatory for others in France, Finland, and Sweden, and strictly mandatory for all game in Italy, Denmark, and Norway (Åhl et al. 2020). Methods for data collection involve e.g., self-reporting online platforms, phone surveys, and field checks, and each method comes with its benefits and challenges in the trade-off between cost and accuracy (Lukacs et al. 2011; Wakeling et al. 2022). Even with mandatory reporting, some non-negligible proportion of hunters typically fail to report (Kahlert et al. 2015; Aubry et al. 2020), and most systems are faced with the challenge of estimating total harvest from partial reporting data.

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In the Swedish system, mandatory reporting is limited to species where hunting is regulated by specific licenses and pre-determined harvest quotas, which includes large carnivores (wolf, bear, lynx, wolverines, seals) and larger ungulates (moose and red deer). With the limitation of hunting season regulations, the harvest of other game species is determined by the land owners or hunters leasing hunting rights. For such game species, i.e., with an open hunting season, the annual harvest is estimated in order to monitor changes in population status and give a basis for management plans. Reporting of these species is voluntary, and the reporting system and estimation of total harvest is administered by the Swedish Association for Hunting and Wildlife Management (SAHWM). Hunting teams report harvest, and reports include the annual harvest of all focal game species as well as their hunting area (in ha) and spatial location to the level of hunting management precinct (HMP, Fig 1).

Estimated harvest is an integral part of game management and monitoring at different scales (local, regional, and national). Some examples include management plans of fallow deer (*Dama dama*) (Sandberg 2014), the incorporation of roe deer (*Capreolus capreolus*) harvest estimates in population models for Lynx (*Lynx lynx*) at different harvest levels (Andrén 2022), and prediction of roe deer and fallow deer browsing damage on commercial forests (Pfeffer et al. 2021). Harvest estimates have also been used to successfully explain shifts in spatial and temporal patterns of three goose species (Liljebäck et al. 2021), as well as effects of climate change (Elmhagen et al. 2015), and responses of the invasive species American mink (*Neovison vison*) to an increased food supply when the population of red fox (*Vulpes vulpes*) was reduced due to an outbreak of sarcoptic mange, *Sarcoptes scabieli*, (Carlsson et al. 2010). Harvest estimates of wild boar (*Sus scrofa*) are used in the works of the Enetwild group, a project run by the European Food Safety Agency (EFSA) and aiming at modeling species distribution and abundance of selected host species and their pathogens Illanas et al. 2022.

Both methods also require information on the total huntable area per HMP. Such data has been made available by SAHWM in previous studies (Lindström and Bergqvist 2020, 2022). However, prior to 2018, after which data was based on estimates of huntable land from Jonsson et al. (2020), the underlying assumptions for the estimation of huntable land were not documented. Efforts to remedy this lack of documentation have been challenged by another issue. Polygon data for HMPs were not stored between 2003 and 2016, and several HMPs have either merged or split during that period. Upon salvaging the spatial data required to re-estimate huntable area, we discovered additional inconsistencies in the polygon data.

This paper has three aims. First, we re-estimate the total huntable area per HMP with documented assumptions. Second, we investigate the implications for estimated harvest by applying the method of Lindström and Bergqvist (2022) to previous and updated estimates of huntable area. Third, because estimates based on linear scaling have been used in several studies and reports, we also compare updated estimates to estimates made with the linear scaling method applied to uncleaned data.

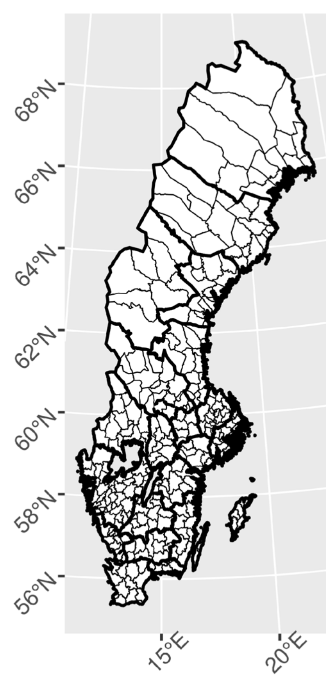


Fig. 1 Updated Hunting Managment Presincts (thin borders) and Counties (thick borders) in Sweden for the hunting year 2021/2022

Methods

Data

Hunting management precincts

Hunting Management Precincts (HMP) are originally a geographical division of SAHWMs members. The HMPs were formed during the early 1980's, and HMP borders originally followed the borders of municipalities or parishes. The number of HMPs has decreased over time, from around 350 to 299 in the hunting year 2021/2022. The most common reason is the merging of two or more adjacent HMPs to create more efficient units and reduce bureaucracy. However, splitting of HMPs also occurs.

Earlier, HMP polygon data were unfortunately not routinely digitally stored. However, as the need for updated digital information became apparent, all HMPs were re-digitalised in 2016 and shapefiles were created and stored annually after that. Also, one shapefile containing 2003 HMP borders was located.

Reporting system

When the system started, hunting teams submitted their harvest and size of their hunting area on paper, and this was entered into a database by SAHWM staff. Now, reports are typically entered directly into the SAHWM-owned online database Viltdata, although other ways of reporting still occur. All reports, around 7,000 per year, are scrutinized by SAHWM staff and, in the case of questions, the reporting person is contacted for clarifications. Typical questions may relate to a species being reported as harvested in an area where the species in question is not normally found or an unusual number of harvested individuals reported (typing error).

Species

The total annual harvest is estimated for all game species in Sweden with an open hunting season as specified in Appendix of the Swedish Hunting Ordinance (SFS 1987:905). The main reason for the estimation is that harvest reporting is voluntary for those species and using only the reported harvest would result in underestimation of the actual harvest. The list of game species where the total harvest is estimated includes a number of species with active management, such as fallow deer (*Dama dama*) and wild boar (*Sus scrofa*).

In this study, we included all species with voluntary reporting except grey partridge, pheasant, and mallards. The reason for their exclusion was that these species are allowed to be released without permits, and unsatisfactory and temporally variable assumptions used in previous harvest estimation methods rendered the results incomparable. The included species are listed in Table 1.

Linear extrapolation method

For HMP k in year t, the LE method estimates the annual harvest as

$$H_{k,t} = \sum_{i\in\mathbb{S}_{k,t}} K_{i,k,t} + \check{H}_{k,t}, \tag{1}$$

where $\mathbb{S}_{k,t}$ is the set of reports for the focal HMP and year, $K_{i,k,t}$ is the reported harvest of team i, and $\check{H}_{k,t} = R_{k,t}\check{A}_{k,t}$ is the estimated harvest on the unreported area. Here, $\check{A}_{k,t}$ denotes the unreported area and $R_{k,t}$ is the estimated harvest per area, which is calculated as

$$R_{k,t} = \begin{cases} \sum_{i\in\mathbb{S}_{k,t}} K_{i,k,t} / \sum_i w_{i,k,t} & \text{if } M_{k,t} \geq 1 \\ \sum_{\kappa\in\mathbb{H}_{l(k)}} \sum_{i\in\mathbb{H}_{\kappa,t}} K_{i,\kappa,t} / \sum_{\kappa\in\mathbb{H}_{l(k)}} \sum_i w_{i,\kappa,t} & \text{if } M_{k,t} = 0 \end{cases}, \tag{2}$$

where, $M_{k,t}$ is the number of reporting teams, $w_{i,k,t}$ is the reported area for team i, and $\mathbb{H}_{l(k)}$ is the set of HMPs in the county in which k is located. County and national estimates are calculated by summing over HMP estimates,

$$C_{l,t} = \sum_{\kappa\in\mathbb{H}_{l(k)}} H_{k,t} \tag{3}$$

and

$$Y_t = \sum_{l\in\mathbb{C}} \sum_{\kappa\in\mathbb{H}_{l(k)}} H_{k,t}, \tag{4}$$

respectively, where $\mathbb{C}$ is the set of counties in Sweden.

The BaHAREHH method

The BaHAREHH method models the expected harvest of team i, covering a proportion $x_{i,k,t}$ of the total huntable area of HMP k at time t, as

$$\nu_{i,k,t} = \mu_{k,t} m_{k,t} \left(\frac{x_{i,k,t}}{\bar{x}_{k,t}} \right)^{\phi_{k,t}}, \tag{5}$$

where $\phi_{k,t}$ models the potentially nonlinear effect of area on hunting rate per team. The division by average proportion of area covered by a team, $\bar{x}_{k,t}$, and multiplication with the average team area, $m_{k,t}$, facilitates the interpretation of $\mu_{k,t}$ as the average hunting rate per area unit of a team with an average area. The reported harvest, $K_{i,k,t}$, is modeled as

$$K_{i,k,t} \sim \text{NegBin}\left(\nu_{i,k,t}, \nu_{i,k,t}\beta_{i,k,t}\right), \tag{6}$$

where $\beta_{i,k,t}$ models the within HMP variability in harvest rate.

Because of the typically sublinear response of harvest to team area, it matters for prediction if the unreported area consists of many small or few large teams. To address the $Q_{k,t}$ number of non-reporting teams, the framework defines for $M_{k,t}$ reporting teams $\mathbf{X} = \left[x_{1,k,t}, x_{2,k,t}, \ldots, x_{M,k,t}, h_{k,t}\right]$, where $h_{k,t}$ is the proportion of huntable area that is unreported, and models

$$\mathbf{X} \sim \text{Dirichlet}(\tilde{\mathbf{a}}) \text{ for } \tilde{a}_i = \begin{cases} a_{k,t} & \text{if } i = 1, 2, \ldots, M_{k,t} \\ a_{k,t}Q_{k,t} & \text{if } i = M_{k,t} + 1 \end{cases}. \tag{7}$$

Here, the concentration parameter, $a_{k,t}$, models variability in area per team for the focal HMP and year.

The BaHAREHH equivalent of Eq. 1 is given by posterior prediction of total harvest of HMP k at time t as

$$H_{k,t} = \sum_i K_{i,k,t} + \int p\left(\check{H}_{k,t} | \boldsymbol{\Xi}_{k,t}, \check{A}_{k,t}\right) p\left(\boldsymbol{\Xi}_{k,t} | \mathbf{D}\right) \mathrm{d}\boldsymbol{\Xi}_{k,t}, \tag{8}$$

where $p\left(\check{H}_{k,t} | \boldsymbol{\Xi}_{k,t}, \check{A}_{k,t}\right)$ is the probability of unreported harvest, $\check{H}_{k,t}$, conditional on HMP specific parameter set $\boldsymbol{\Xi}_{k,t} = \left[Q_{k,t}, a_{k,t}, \mu_{k,t}, \phi_{k,t}, \beta_{k,t}\right]$ and unreported huntable area,

$\breve{A}_{k,t}$, and $p\left(\boldsymbol{\Xi}_{k,t}|\mathbf{D}\right)$ is the posterior distribution of the parameter set conditional on all available data, $\mathbf{D}$.

To reduce parameter uncertainty through Bayesian shrinkage, the framework includes a hierarchical model structure with random effects at the county and HMP levels. Denoting with $\theta_{k,t}$ a focal parameter of $\Xi_{k,t}$,

$$\log\left(\theta_{k,t}\right) = \omega_{\theta,t} + \lambda_{\theta,l(k),t} + \chi_{\theta,k,t}. \tag{9}$$

Here, $\omega_{\theta,t}$, $\lambda_{\theta,l(k),t}$, and $\chi_{\theta,k,t}$ (the latter of which is omitted for modeling of $\theta = a$, ϕ, and β) are the nationwide, county, and HMP level effects, respectively, with $l(k)$ indicating the county in which HMP k is located. Further, autoregressive modeling is applied at the national, county, and HMP levels by defining

$$\begin{aligned} &\omega_{\theta,t} \sim \text{Normal}\left(\omega_{\theta,t-1}, \sigma_{\omega,\theta}\right) \text{ if } t > 1 \\ &\lambda_{\theta,l(k),t} \sim \begin{cases} \text{Normal}\left(0, \sigma_{\lambda,\theta}\right) \text{ if } t = 1 \\ \text{Normal}\left(\rho_{\lambda,\theta}\lambda_{\theta,l(k),t-1}, \left(1-\rho^2_{\lambda,\theta}\right)\sigma_{\lambda,\theta}\right) \text{ if } t > 1 \end{cases} \\ &\chi_{\theta,k,t} \sim \begin{cases} \text{Normal}\left(0, \sigma_{\chi,\theta}\right) \text{ if } t = 1 \\ \text{Normal}\left(\rho_{\chi,\theta}\breve{\chi}_{\theta,k,t-1}, \left(1-\rho^2_{\chi,\theta}\right)\sigma_{\chi,\theta}\right) \text{ if } t > 1. \end{cases} \end{aligned} \tag{10}$$

The parameters $\sigma_{\omega,\theta}$, $\sigma_{\lambda,\theta}$, $\sigma_{\chi,\theta}$, $\rho_{\lambda,\theta}$, and $\rho_{\chi,\theta}$ define spatiotemporal autcorrelation at the levels of interest. The parent node for the national level parameter at the index year, $\omega_{\theta,1}$, is the prior. The notation $\breve{\chi}_{\theta,k,t-1}$ is used when defining a model for HMPs that change between years. These and other details are available in Lindström and Bergqvist (2022).

Computation of posteriors was executed with Stan (Carpenter et al. 2017), which uses Hamiltonian Monte Carlo methods to sample from the posterior distribution. We ran four chains, each with 2000 iterations, the first 1000 of which was used for burn-in. A minority of analyses exhibited divergent transitions with Stan's default sampling parameters. To circumvent this, we reran the sampler with gradually increased targeted average acceptance probability (adapt_delta, default 0.8) until no divergent transitions occurred. The highest implemented adapt_delta was 0.99 and was required for two analyses: *Anser albifrons* and *Melanitta nigra*, in both cases for analyses based on previous estimated huntable land. We used potential scale reduction factor (PSRF) and effective sample size (ESS) Gelman et al. (2004) to assess computational efficiency. Additional computational details are described in Lindström and Bergqvist (2022). The code is available from https://github.com/tomli071/BaHAREHHpub.

Making polygon data consistent

The available polygon data had several within- and between-year issues. Within-year issues included missing HMP IDs, merged HMPs still included as defunct HMP polygons, overlaps between HMPs, and terrestrial areas not encompassed by any HMP. There were also instances of substantial mismatches between the huntable land in the spreadsheet data used for estimation and the polygon data as the result of human errors when merging polygons or entering data into the spreadsheet. Between-year issues included a lack of polygon data for the years 2004–2015 and inconsistencies in the available files such that the polygon differed for the same HMP in different years. Consequently, the union of all HMPs was not constant, and these inconsistencies were most substantial when comparing the 2016 and 2003 polygons.

We describe the details of making the polygons consistent in Appendix. Figure 2 exemplifies the updating of spatial data with the county of Västra Götaland. All cleaned 2021 HMP polygons were joined with base maps, showing e.g. roads and towns, and sent to local representatives of SAHWM to ensure the cleaned data corresponded to the real borders and that no remaining issues could be identified.

Estimating huntable land

Previous estimates

For HMP data prior to 2018, there was no documentation available for the assumptions used to estimate huntable land. However, the estimation was based on official data and water as well as large infrastructure were excluded. When HMPs were changed, their respective huntable areas were added to the new HMPs in proportion to their respective polygon area, i.e. no new estimation of huntable land was performed.

For the 2018 HMP data, huntable land was re-estimated in Jonsson et al. (2020). The procedure was done in two steps. First, areas where hunting is forbidden were excluded. These include national parks (Swedish Environmental Protection Agency 2023) and state-owned land above the cultivation limit (i.e. the high mountains). In the latter case, hunting is allowed but all harvest should be reported to the County Administrative Board and is therefore not included in the estimation of total harvest. For the remaining areas, the land cover was estimated from (Swedish Environmental Protection Agency 2020), which provides raster data of estimated land cover in 10x10 m^2 grids based on satellite information. There are 25 thematic classes, which are grouped into six higher-order classes: forest, open wetland, arable land, other open land, artificial surfaces, and water. Of these, the first four and water within 25 m of land were classified as huntable. Estimated huntable land (EHL) in subsequent years has been based on the estimates provided in Jonsson

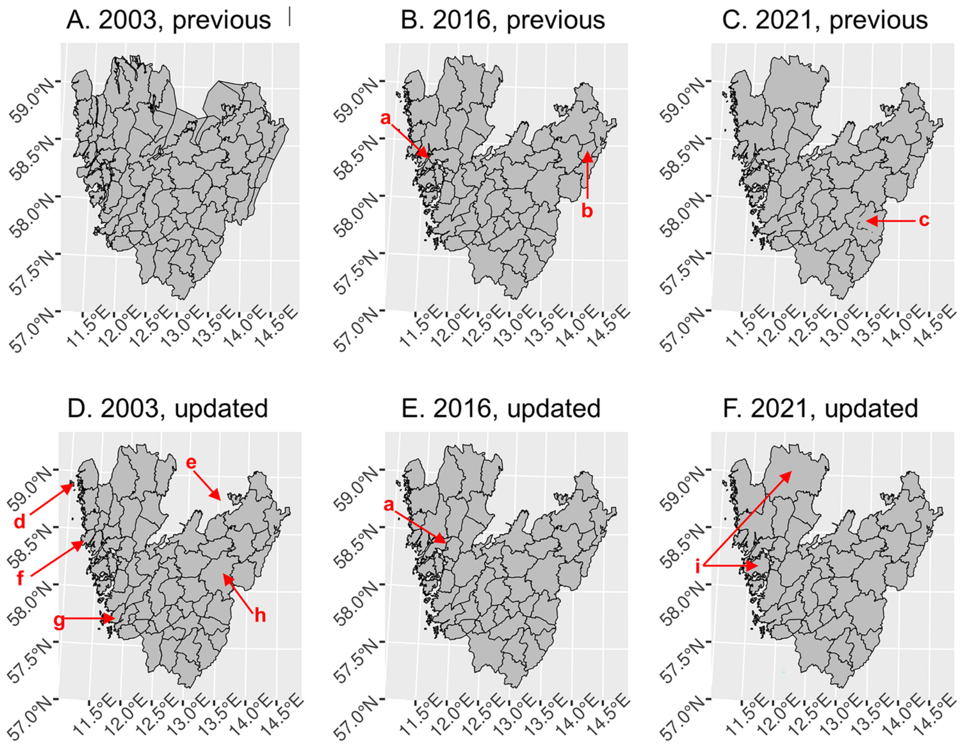


Fig. 2 Updated and previous hunting management precincts (HMPs) of the county Västra Götaland for 2003, 2016, and 2021, exemplifying the process of cleaning and standardizing the HMP polygons. Arrow **a** indicates a major change in the polygons of HMPs Bokenäset and Uddevallanejden, where the wrong polygons had been joined in the previous data after HMP Bokenäs–Skaftö, merged with Skredsvik in 2012. This error was amended by using the borders of the 2003 data while maintaining the union of the involved 2016 HMPs. Arrow **b** indicates Tibro which was split, and the parts were merged with three adjacent HMPs. In the updated polygons, the union of the involved HMPs remained unchanged. Arrow **c** indicates that, after Hökerum merged with Åsunden in 2018 and subsequently with Redsväg in 2020, the joined geometry included lines and minor gaps excluded from the HMP in the uncleaned polygon data. These were removed in the updating process. Arrows **d** and **e** show that island and water bodies (here lake Vänern) included in the HMP polygons were made consistent. Arrow **f** indicates an HMP Sotenäs, which has remained unchanged 2003–2021 and therefore has the same polygon for all years in the updated data. Arrow **g** indicates an example where two HMPs (Göteborg and Hisingen) were merged to Hisingen–Göteborg, in 2006. The polygons of 2003 were used to define borders prior to the merge while maintaining the union equal to Hisingen–Göteborg. Arrow **h** indicates HMP Falbyggden, which was included as three disjoint polygons in the previous polygon data but was updated to the 2016 polygon. Arrow **i** indicates HMPs Orust, which was formed by merging Västra Orust and Östra Orust in 2019, and Norra Dal, which was formed by merging of Bengtsfors, Dals, and Åmål, in 2019. In the spreadsheet data, the huntable area of Bengtsfors, was erroneously added to Orust instead of Norra Dal

et al. (2020), assuming that huntable land is proportional to the area of the parts when HMPs are split.

The updated Polygons are available in the associated data publication (Reserved DOI: 10.17632/f37cbghz78.2).

Updating estimated huntable land

Using the updated HMP polygon data, we here re-estimated huntable land for every HMP and year. We applied the same rules as in Jonsson et al. (2020) to define EHL based on data from Swedish Environmental Protection Agency (2020), with the exception that all water was excluded. The reason was to make the estimation coherent with other Swedish monitoring systems, specifically the moose management system operated by the County Administrative Boards, in which huntable land is estimated excluding water.

New national parks were formed during the period 2003–2021, and we excluded the area of the national park if the start date occurred before or within the end of the year. Because the hunting year runs from July first to June 30, the national park was excluded if it affected hunting for more than 50% of the period.

Five HMPs in Stockholm county (Solna Sundbyberg, Danderyd, Lidingö, Västerort, and Stockholm Centrala) and one in Uppsala county (Uppsala) are inner-city HMPs where no hunting occurs, and the areas of these polygons were removed from huntable land. Danderyd was incorporated into Norrort in 2018, but the EHL was kept constant because the area of Danderyd was excluded.

The terra package, version 1.7-3, was used to crop the land cover raster by HMP polygons.

Metrics for comparison of estimated harvest

To investigate how harvest estimates depend on methodology and EHL, we estimated the harvest of the included species from 2003 to 2021 with three combinations of methods and data: BaHAREHH with updated EHL (denoted BaHAREHH^{+}), BaHAREHH with previous EHL (denoted BaHAREHH^{-}), and LE with previous EHL (denoted LE^{-}). The BaHAREHH estimates provide posterior predictive distributions, and, to make comparisons with the point estimate of the LE method, we focused on the median harvest estimates, indicated in the below notation with the ~ accent. For LE^{-}, this is equal to the point estimates of Eqs. 1, 3, and 4. Using superscripts to indicate estimation method and M indicating either BaHAREHH^{-} or LE^{-}, we calculated

$$
\begin{aligned}
r_t^{\{\mathrm{Y,M}\}} &= \frac{\tilde{Y}_t^{\{\mathrm{M}\}}}{\tilde{Y}_t^{\{\mathrm{BaHAREHH}^+\}}} \\
r_{l,t}^{\{\mathrm{C,M}\}} &= \frac{\tilde{C}_{l,t}^{\{\mathrm{M}\}}+1}{\tilde{C}_{l,t}^{\{\mathrm{BaHAREHH}^+\}}+1} \\
r_{k,t}^{\{\mathrm{H,M}\}} &= \frac{\tilde{H}_{k,t}^{\{\mathrm{M}\}}+1}{\tilde{H}_{k,t}^{\{\mathrm{BaHAREHH}^+\}}+1}.
\end{aligned}
\tag{11}
$$

The +1 for county and HMP estimates was implemented to prevent zeros in the denominators. To avoid results dominated by areas where no hunting of a focal species occurs, we excluded for further analyses counties and/or HMPs in which all methods estimated null-harvest. For HMP and county comparisons, we pooled all combinations of year and the focal level into a single distribution of ratios. As a summary statistic to describe the spread of the ratio distributions, we calculated the geometrical standard deviation, $\Sigma_{\mathrm{geo}} = \exp(\mathrm{SD}(\log(\mathbf{r})))$, where SD indicate the arithmetic standard deviation.

At the annual level, we also considered three higher-order statistics of the time series $\tilde{\mathbf{Y}}^{\{\mathrm{M}\}} = \tilde{Y}_{2003}^{\{\mathrm{M}\}}, \tilde{Y}_{2004}^{\{\mathrm{M}\}}, \ldots, \tilde{Y}_{2021}^{\{\mathrm{M}\}}$, where M indicates any of the three estimation methods. First, to investigate any effect on temporal trends, we fit with least-square regression a linear model, $B_0 + B_1 t$, and compared the slope, B_1, across the estimates. Second, we compared temporal variability, quantified by Coefficient of variation, $\mathrm{SD}\left(\tilde{\mathbf{Y}}^{\{\mathrm{M}\}}\right)/\mathrm{mean}\left(\tilde{\mathbf{Y}}^{\{\mathrm{M}\}}\right)$. Third, we compared temporal autocorrelation, quantified by Pearson correlation between $\tilde{Y}_{2004}^{\{\mathrm{M}\}}, \tilde{Y}_{2005}^{\{M\}}, \ldots, \tilde{Y}_{2021}^{\{\mathrm{M}\}}$ and $\tilde{Y}_{2003}^{\{\mathrm{M}\}}, \tilde{Y}_{2004}^{\{\mathrm{M}\}}, \ldots, \tilde{Y}_{2020}^{\{\mathrm{M}\}}$.

The estimated harvest at HMP, count, and total annual harvest are available in the associated data publication (Reserved DOI: 10.17632/f37cbghz78.2).

Results

Effects on estimated Huntable Land

The updated total EHL decreased on average (over years) by 0.17% compared to the previous. The previous average was 33,152,305 ha, ranging from 33,079,181 ha in 2011–2019 to 33,683,651 in 2020–2021. The updated average was 33,094,743 ha, ranging from 33,094,311 ha in 2018–2021 to 33,094,947 in 2003–2008.

At the HMP level, the updated EHL also corresponded well with the previous, both for early and later years (Fig 3). There were however HMPs where the updating of EHL made a substantial difference. The largest relative increase in EHL following the updating was found for HMP 730, Bokenäset, where EHL increased by 317%, for 2020–2021. The largest relative decrease was found for HMP 836, Orust, which decreased to 27% and 28% of its previous EHL for 2019 and 2020–2021, respectively.

Effects on harvest estimates

The average ratio between BaHAREHH applied to previous and updated EHL (Fig. 4A), $\mathrm{mean}\left(\mathbf{r}^{\{\mathrm{Y,BaHAREHH}^-\}}\right)$, ranged from 4.1% lower for *M. penelope* to 1.0% higher for *L. timidus*. Differences for individual years ranged from 22.5% lower for *M. nigra* in 2006 to 10.0% higher for *M. nigra* in 2007. Estimates of $\Sigma_{\mathrm{geo}}^{\{\mathrm{Y,BaHAREHH}^-\}}$ ranged from 1.00 for *L. timidus* to 1.05 for *M. penelope*.

The average national harvest based on LE^{-} applied to previous EHL ranged from 19.5% lower for *A. albifrons* to 54.2% higher for *M. nigra* than BaHARREH^{+}. Differences

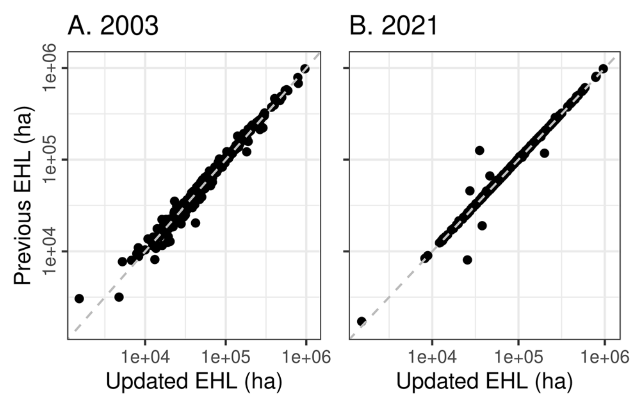


Fig. 3 Previous vs. updated estimated huntable land (EHL) per hunting management precinct in **A** 2003 and **B** 2021. The dashed line indicates equal EHL

for individual years ranged from 99% lower for *A. albifrons* in 2008 lower to 408% higher for*M. nigra* in 2016. Estimates of $\Sigma_{\text{geo}}^{\{\text{Y,LE}^-\}}$ ranged from 1.02 for *C. capreolus* to 4.18 for *M. nigra*.

At the county and HMP levels, extremes were largely driven by low harvest estimates where at least one of the methods (typically LE) was estimated at zero, and we focus on Σ_{geo} to highlight differences between methods. For comparison of BaHAREHH applied to previous and updated EHL, $\sigma_{\text{geo}}^{\{\text{C,BaHAREHH}^-\}}$ ranged from 1.02 for *A. albifrons* to 1.21 for *M. penelope* and $\Sigma_{\text{geo}}^{\{\text{H,BaHAREHH}^-\}}$ from 1.03 for *M. nigra* to 1.57 for *S. scrofa*. For the corresponding comparison to LE applied to the previous EHL, $\Sigma_{\text{geo}}^{\{\text{C,LE}^-\}}$ ranged from 1.13 for *C. capreolus* to 3.54 for *C. frugilegus* and $\Sigma_{\text{geo}}^{\{\text{H,LE}^-\}}$ from 1.97 for *M. nigra* to 7.93 for *C. frugilegus* at the HMP level.

Figure 5A shows the estimated temporal trend for BaHAREHH$^+$, BaHAREHH$^-$, and LE$^-$ harvest estimates. Though differences were minor and barely visible, slopes were higher for BaHAREHH$^-$ estimates than for BaHAREHH$^+$ for all species, and the largest difference was observed for *A. anser*, for which BaHAREHH$^-$ was 0.0033 year^{-1} higher. Slopes for LE$^-$ were higher than for BaHAREHH$^+$ in 13 of 34 species, and the largest difference was observed for *M. nigra*, for which the LE$^-$ slope was 0.034 year^{-1} larger. Fig. 5A reveals a pattern such that the slope for LE$^-$ is typically larger than the slope for BaHAREHH$^+$ when the latter is positive and the reversed relationship is found for negative slopes of BaHAREHH$^+$. With 27 out of 34 species following this pattern, the trend is significantly non-random in a binomial test with p=0.0008.

The coefficient of variation (Fig 5B) was higher for BaHAREHH$^-$ estimates than for BaHAREHH$^+$ in 13 of 34 species, and the largest difference was observed for *M. nigra*, for which CoV was 0.016 higher. For LE$^-$ estimates, CoV was higher than for BaHAREHH$^+$ estimates in 33 species, and the largest difference was observed for *M. nigra*, for which CoV was 0.683 higher.

Temporal autocorrelation (Fig 5C) was lower for BaHAREHH$^-$ estimates than for BaHAREHH$^+$ in 21 species. Again, these differences were minor, and the largest difference was observed for *V. vulpes*, for which BaHAREHH$^-$ AC was 0.025 lower. For LE$^-$ estimates, AC was lower than for BaHAREHH$^+$ estimates in all 34 species, and the largest difference was observed for *P. pica*, for which AC was 0.63 lower.

Computation

Across all analyses, the highest PSRF was observed for one of the $\chi_{\mu,k,t}$ parameters for *D. dama* and updated EHL at 1.03. The lowest ESS was observed for the $\sigma_{\lambda,\mu}$ for *A. albifrons* at 209. The latter could be considered low, depending on the application. Fortunately, diagnostics for posterior predictive samples of $H_{k,t}$, the estimated harvest at the HMP levels (which are also used to define higher level harvest estimates), consistently exhibited satisfying diagnostics. The average ESS was close to the theoretical maximum of 4000 for all species and analyses, and the minimum ESS observed was 2190 (for one HMP of *L. muta*). All PSRF estimates for $H_{k,t}$ samples were ≤ 1.01

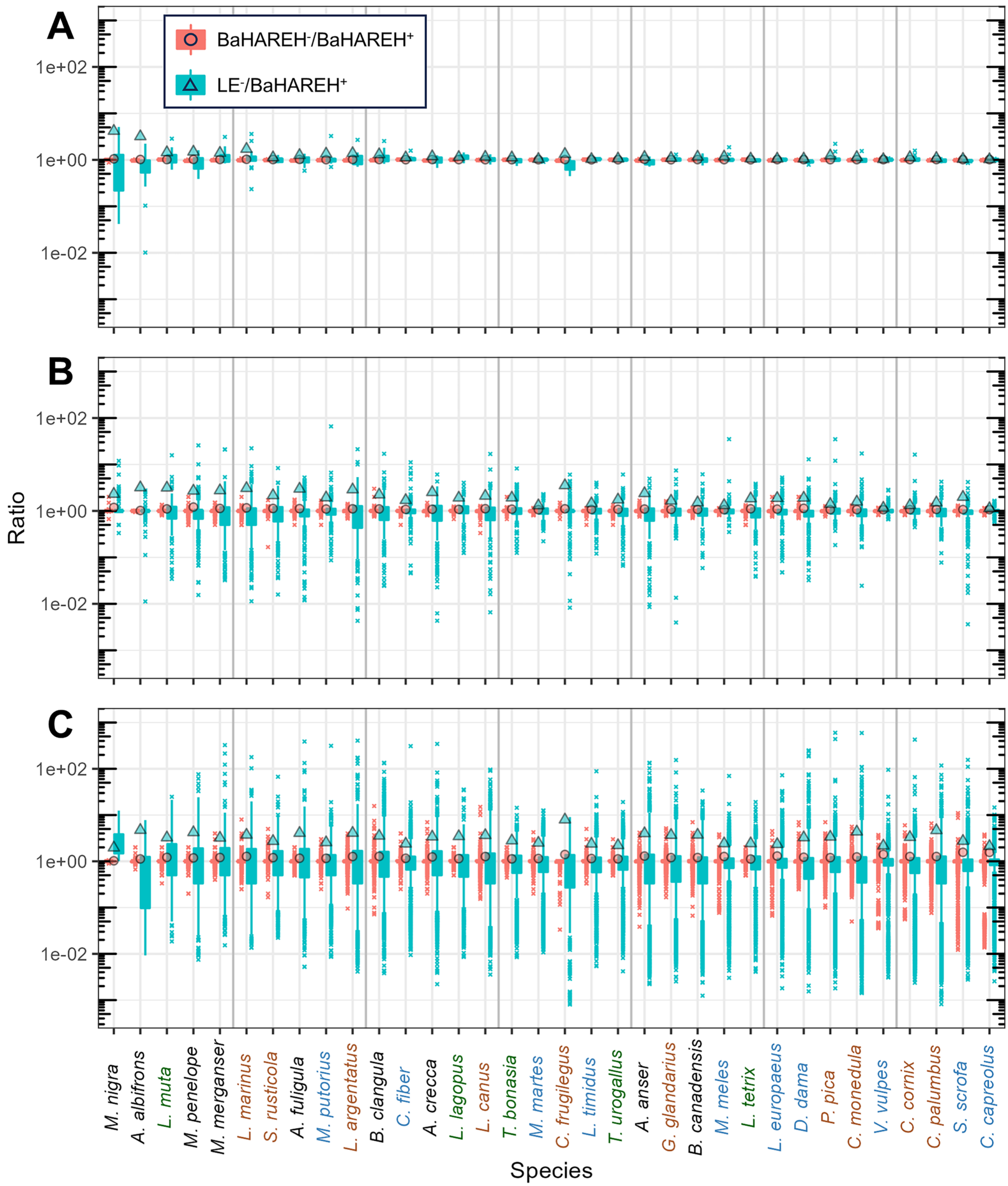
Discussion

Starting in 1995, LE and previous EHL were used in Sweden's official harvest statistics for open season game. Since these statistics have been used extensively in wildlife research and management, we here investigated how the switch to BaHAREHH and updated spatial data affects the harvest estimates. We also applied BaHAREHH to the previous EHL to investigate if the updating of EHL itself can explain observed differences.

We updated HMP polygons and re-estimated EHL to solve the issue of erroneous, undocumented, and inconsistent EHL. After updating, the total EHL only differed between years as a result of the formation of new national parks. At the national level, there were small differences between previous and updated EHL. At the HMP level, updated EHL generally corresponded well with previous (Fig. 3), though substantial differences were observed.

These differences were corroborated by the results of applying BaHAREHH to updated and previous EHL. Annual harvest estimates (Fig. 4A) and time-series statistics (Fig. 5) at the national level were highly similar, even if the slope of the temporal trend was larger for BaHAREHH$^-$ than BaHAREHH$^+$ in all but one species. The magnitude of differences was small (an order of magnitude smaller than for comparison with LE$^-$) and can be explained by the fact that, when huntable land was updated with standardized assumptions across years, EHL was reduced for the later years, primarily because water was removed. This choice was made to make the assumptions similar to other reporting systems and promote comparisons across species and reporting systems. Most importantly, the updated EHL (which will henceforth be used in harvest estimation of game with open hunting season in Sweden) was derived with the same (documented) assumptions, and any observed trends are due to the reports, not different assumptions regarding the area for which unreported harvest must be predicted.

County and HMP harvest estimates for BaHAREHH$^-$ and BaHAREHH$^+$ were also typically similar, but distinct differences were observed, in particular for individual HMPs. For instance, the box and whiskers (capturing the bulk of the distribution of ratios) are close to zero for all species (Fig. 4C).



Yet, $\Sigma_{\text{geo}}^{\{\text{H,BaHAREHH}^{-}\}}$ indicated at least moderate discrepancy for some species. This statistic is interpreted as what the log-ratio at one standard deviation from the mean corresponds to in terms of ratio. A ratio of $\Sigma_{\text{geo}}^{\{\text{H,BaHAREHH}^{-}\}} = 1.57$ (*S. scrofa*) is considerable and is primarily the result of outlier HMPs for which the updating of EHL had a substantial effect.

The ratios between the two methods were centered around zero at all levels (Fig. 4), indicating that there is

◀**Fig. 4** Ratio of harvest estimates based on either Bayesian Hierarchical Autoregressive Estimation of Hunting Harvest (BaHAREHH) or linear extrapolation applied to previous estimated of huntable land (EHL), denoted BaHAREHH^{-} and LE^{-}, respectively, to BaHAREHH estimates applied to updated EHL, denoted BaHAREHH^{+}. Estimates were compared at the national (**A**), county (**B**), and hunting management precinct (HMP) (**C**) levels. Boxplots (boxes and whiskers indicating central 50% of the distribution and 1.5 times the inter-quartal distance, respectively) show the distribution of ratios between of yearly estimates (2003–2021) or the combination of year and county/HMP. Circles and triangles indicate the geometric standard deviation of ratios. Species are sorted from left to right according to ascending mean national BaHAREHH^{+} estimates, and text colors indicate taxonomic groups mammals (blue), waterfowl (black), grouse (green), and other birds (brown)

no apparent bias in the previous estimates compared to BaHAREHH^{+}. However, even at the level of total annual harvest (Fig. 4 A), there are substantial differences for individual years. Slopes fitted to annual LE^{-} estimates were typically amplified (Fig 5A), suggesting that when previous harvest estimates have been used as an indicator of rapidly increasing or decreasing population trends, these trends may have been exaggerated. Further, the autocorrelations were lower and coefficients of variation higher (Fig. 5B and C). These results are expected when adding random noise to time series, which the LE method is prone to because of the sensitivity to individual reports.

At the county and, in particular, the HMP levels, there were more substantial differences between LE^{-} and BaHAREHH^{+} estimates. The $\Sigma_{\mathrm{geo}}^{\{C,LE^{-}\}}$ ranged from 1.13 to 3.54 and $\Sigma_{\mathrm{geo}}^{\{H,LE^{-}\}}$ from 1.97 to 7.93, indicating that there are generally large discrepancies between estimates at the county and, in particular, HMP levels. For most species, some ratios differed by more than one and two orders of magnitude at the county and HMP level, respectively (4 B and C). Thus, the sensitivity to low reporting in the LE method is more pronounced at the lower levels.

There is no complete harvest data available, making it impossible to assess exactly which model is closest to the true harvest. Yet, cross-validation studies and model checks (Lindström and Bergqvist 2020, 2022) have shown that BaHAREHH, which implements borrowing of strength in time and space and acknowledges within-HMP variability and non-linear relationship between area and harvest rate, improves predictive performance and reduce the sensitivity to low reporting. Thus, it is safe to say that BaHAREHH estimates are more reliable than LE. Previous harvest estimates have been used at the national (e.g., Liljebäck et al. 2021; Lozano et al. 2023; Heldbjerg et al. 2019), county (e.g., Thulin et al. 2021; Carlsson et al. 2010) and HMP (e.g. Elmhagen et al. 2015; Aronsson et al. 2016) levels. Some studies have aggregated several counties into regions of interest (Andrén 2022) or used overlap with HMPs to define harvest at other levels, such as Moose Management Area (Neumann et al. 2020; Pfeffer et al. 2021 or municipality Neumann et al. 2022).

The potential impact of the estimation method depends on the question, species, and scale of interest. Ungulate populations are of particular interest in wildlife management because of their importance for hunting (Wiklund and Malmfors 2014) and consequences for society. These include collisions with vehicles (Gren and Jägerbrand 2019) and trains (Trafikverket 2015) and damage to agriculture (SCB 2015; Menichetti et al. 2019; Gren et al. 2020) and forestry (Månsson and Jarnemo 2013; Sjölander-Lindqvist and Sandström 2019; Pfeffer et al. 2021). In particular *S. scrofa* is also a wildlife reservoir for many pathogens (Wallander et al. 2015; Stenberg et al. 2022; Ernholm et al. 2022), and a recent outbreak of African Swine Fever Jordbruksverket 2023 has highlighted the importance of reliable surveillance data for the species. Fortunately, *C. capreolus*, *D.dama*, and *S. scrofa* harvest estimates were similar at the level of total annual harvest, suggesting that previous conclusions (based on LE^{-}) at this level are robust to the estimation method.

Thulin et al. (2021) studied temporal changes in hare (*Lepus europaeus* and *L. timidus*) and red fox (*V. vulpes*) at both national and county levels. Like the ungulates, these species were largely insensitive to the estimation method (Figs. 4 and 5) at the national level, and general trends are likely not a result of the estimation method, yet identified differences between individual years at the county level may be less robust.

Neumann et al. (2020) studied the relationship between harvest estimates and ungulate-vehicle collisions (UVCs) at the level of moose management areas (MMA). The study found a strong relationship between harvest and UVCs, but the relationship was weaker for fallow deer than for wild boar and roe deer. Pfeffer et al. (2021) identified a significant effect of roe deer harvest on winter damage to pine stands, but not for fallow deer. There are on average ~2.5 HMPs per MMA, and we may only speculate if and to what extent the sensitivity to individual reports of the LE methods, which is more evident for species with lower harvest, may have influenced such results. Yet, the additional randomness to harvest estimates may indeed weaken or mask relationships to other data sources. Also, the issues identified for some HMP borders (e.g. Fig. 2) may also affect results when these are used to define overlap with other spatial units.

Certainly, any implications of the precariousness of previous harvest estimates (particularly at the HMP level) are no shortcomings of the authors of the studies that have used them. Large-scale studies of wildlife are inherently challenging and must rely on the indirect proxies of populations that are available. Harvest data is crucial to wildlife management, and currently, the BaHAREHH framework provides the most reliable estimates for the Swedish reporting system. This study has aimed to provide an overview of the BaHAREHH

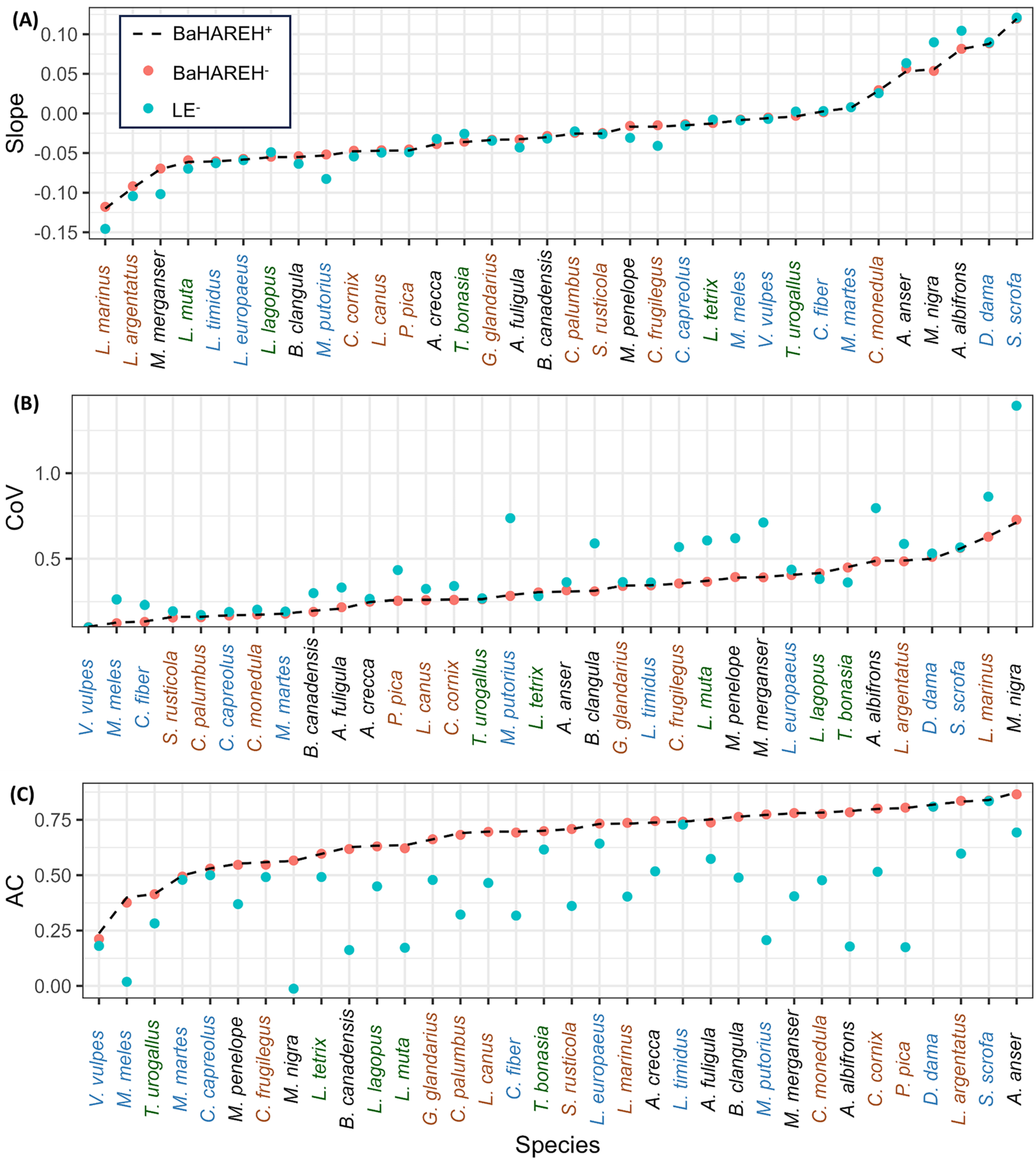


Fig. 5 Comparison of slope (panel **A**), autocorrelation (AC, panel **B**), and coefficient of variation (CoV, panel **C**) of time series of estimated annual harvest, 2003–2021. Dashed lines indicate metrics based on the Hierarchical Autoregressive Estimation of Hunting Harvest (BaHAREHH) method applied to updated estimated huntable land (EHL), denoted (BaHAREHH^{+}), and colored markers indicate metrics of BaHAREHH or linear extrapolation (LE) applied to previous EHL, denoted BaHAREHH^{-} and LE^{-}, respectively. Species are sorted according to the BaHAREHH^{+} metric, and text colors indicate taxonomic groups: mammals (blue), waterfowl (black), grouse (green), and other birds (brown)

Table 1 Included species and estimated annual average harvest based on the Bayesian Hierarchical Autoregressive Estimation of Hunting Harvest method applied to updated Estimated Huntable Land

Species	Common name	Taxonomic group	Average harvest
Lagopus lagopus	Willow ptarmigan	Grouse	7,319
Lagopus muta	Rock ptarmigan	Grouse	743
Lyrurus tetrix	Black grouse	Grouse	27,417
Tetrao urogallus	Western capercaillie	Grouse	21,703
Tetrastes bonasia	Hazel grouse	Grouse	8,699
Capreolus capreolus	Roe deer	Mammal	113,446
Castor fiber	Beaver	Mammal	6,855
Dama dama	Fallow deer	Mammal	35,592
Lepus europaeus	European hare	Mammal	33,972
Lepus timidus	Mountain hare	Mammal	20,738
Martes martes	European pine marten	Mammal	9,580
Meles meles	Eurasian badger	Mammal	26,673
Mustela putorius	European polecat	Mammal	2,932
Sus scrofa	Wild boar	Mammal	80,553
Vulpes vulpes	Red fox	Mammal	65,707
Columba palumbus	Common wood pigeon	Other bird	72,384
Corvus cornix	Hooded crow	Other bird	71,319
Corvus frugilegus	Rook	Other bird	11,797
Corvus monedula	Western jackdaw	Other bird	59,995
Garrulus glandarius	Eurasian jay	Other bird	24,440
Larus argentatus	European herring gull	Other bird	6,193
Larus canus	Common gull	Other bird	8,524
Larus marinus	Great black-backed gull	Other bird	1,381
Pica pica	Eurasian magpie	Other bird	43,645
Scolopax rusticola	Eurasian woodcock	Other bird	1,393
Anas crecca	Eurasian teal	Waterfowl	7,072
Anser albifrons	Greater white-fronted goose	Waterfowl	300
Anser anser	Greylag goose	Waterfowl	22,036
Aythya fuligula	Tufted duck	Waterfowl	2,589
Branta canadensis	Canada goose	Waterfowl	26,440
Bucephala clangula	Common goldeneye	Waterfowl	6,748
Mareca penelope	Eurasian wigeon	Waterfowl	1,118
Melanitta nigra	Common scoter	Waterfowl	40
Mergus merganser	Common meganser	Waterfowl	1,192

method and provide documented and transparent analyses of huntable land for which harvest must be estimated. We anticipate that the presented analyses will promote an understanding of at what levels and for which species the switch to BaHAREHH may change harvest estimates compared to previously available statistics.

Appendix

We started by solving any within-year issues for 2016. We used this as our reference year because any changes to the HMPs in later years were based on the 2016 polygon data, and there were more issues with the 2003 data, particularly that several IDs were missing and many islands were excluded. Missing or incorrect HMP IDs were added/corrected. HMP Trollhättan Lilla Edet was missing, but the HMPs that should have been merged (Lilla Edet and Trollhättan) were included. Thus, the Trollhättan Lilla Edet polygon was changed to the union of these defunct HMPs.

HMP Bokenäset was suspiciously small with a total area of 8600 ha compared to 29000 ha huntable area listed in the estimation spreadsheet. Concurrently, HMP Uddevallanejden was suspiciously large. When HMPs Bokenäs–Skaftö and Skredsvik merged in 2008, the polygon of Skredsvik had erroneously been added to HMP Uddevallanejden. To amend this issue, we used the 2003 polygon data (from before the merge), and the intersection between Uddevallanejden and defunct Skredsvik was removed from Uddevallanejden and instead added to Bokenäset.

A similar issue, which also affected between-year consistency, was identified for HMPs in the county Halland. Here, HMPs Hishult and Knäred, merged in 2012 to form Södra Halland. Also, Lagadalen and Veinge merged in 2014 to form Höks. From the documentation, it was unclear which polygons in 2003 should constitute which defunct HMPs. Also, comparing polygon areas with the EHL of the estimation data, Södra Halland was too large and Höks was too small, and the differing area corresponded to approximately one of the polygons in the 2003 data. Based on the documentation of huntable area, the assumption that merging HMPs must be adjacent, and the names of defunct HMPs, which often correspond to towns and places in the area, we were able to recreate plausible polygons prior to each merge. These were communicated with local SAHWM representatives, who confirmed that these were the most likely scenarios. Yet, due to staff turnover since previous merges, there remains some uncertainty regarding previous borders between these defunct HMPs.

There were 13 HMPs that overlapped in the 2016 polygon data. For these HMPs, we choose to remove the intersecting area from the HMP with the lower index. The indices are arbitrary, and we applied this approach rather than randomly selecting which of the overlapping HMPs to crop to not introduce randomness into the process. The largest overlap was 287 ha, which reduced the area of HMP Jokkmokk with 0.98%.

To achieve between-year consistency, we let the updated polygons from 2016 define polygons for HMPs that had not changed in previous or subsequent years. For HMPs that had changed between 2003 and 2016, we were able to identify from the documentation which polygons in the 2003 data defined the HMPs in 2003 and used these to define borders of defunct HMPs. To ensure that the union of the defunct HMPs remained equal to the polygons HMPs they had merged into in 2016, 2003 polygons were cropped by the 2016 polygon, and any area not covered by the resulting polygons was added to the closest defunct HMP. If the distance was identical (typically areas located as a gap between two HMPs), the non-covered area was added to the HMP with lower ID. For overlaps between HMP polygons, we cropped the HMP with the lower ID by the intersection. We were able to recreate consistent HMP data from 2004–2015 by either copying the polygons from the 2016 data or joining polygons from the previous year based on documentation.

Similarly, we replaced polygons in the data for 2017–2021 by the cleaned 2016 polygons or by sequentially merging HMPs from the previous year. There were instances where an HMP split into two or more parts and the parts were merged with different HMPs. In these cases, we let the available polygon data define the borders and cropped and added uncovered areas with the same rules as when cleaning the 2003 data to ensure the union of the involved HMPs remained unchanged.

All polygon manipulations were performed with the sf package, version 1.0-12, using SWEREF99 projections.

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Author contributions TL and GB designed the study and wrote the main manuscript. TL, PJ, and FS updated spatial data and estimated huntable land with documented assumptions. TL estimated harvest with all methods, analyzed results, and prepared figures. All authors reviewed the manuscript.

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Data availibility statement Harvest estimates and updated polygon data have been deposited in Mendeley Data, doi: 10.17632/f37cbghz78.2.

Declarations

Ethical approval No animal or human trials were used in the study. Harvest reports were accessed without any personal information.

Competing interests The authors have no competing interests as defined by Springer, or other interests that might be perceived to influence the results and/or discussion reported in this paper.

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