Relative abundance of select reef fishes from Southeast Reef Fish Survey video data, 2011–2023

Nathan M. Bacheler[1] and Walter Bubley[2]

[1]Southeast Fisheries Science Center, National Marine Fisheries Service, Beaufort, NC 28516
[2]South Carolina Department of Natural Resources, Marine Resources Research Institute, Charleston, SC 29412

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Introduction

Reef fish species along the southeast United States Atlantic coast (hereafter, SEUS) have been monitored by the Marine Resources Monitoring, Assessment, and Prediction program (MARMAP) and the Southeast Reef Fish Survey (SERFS) with chevron traps since 1990. These trap data have been central components of the stock assessments of many various fish species in the SEUS, and trap indices for a number of species have been summarized annually by the South Carolina Department of Natural Resources (e.g., Vecchio et al. 2024). Beginning in 2011, video cameras were attached to all traps deployed region-wide by SERFS to provide additional information about reef fish relative abundance and seafloor habitat. The goal of this report is to provide video-based indices of abundance for many reef fish species in the SEUS, which complements the summary of trap-based indices of abundance of Vecchio et al. (2024).

Caveats

- Video-based indices of abundance are not an indication of stock status, the latter of which requires additional information such landings, length and age compositions, and life history parameters.
- The ways in which video-based indices of abundance are standardized in this report (e.g., model selection, predictor variables included) may be different from those used in SEDAR stock assessments.
- Video reading takes many months to complete, often finishing approximately 1 year after collection. Thus, the last year of video data included in this report is 2023.
- Species were only included here if the SERFS video index was included in a recent SEDAR assessment or the CVs were low enough that the video index might be useful.

Results

A total of 15,507 videos were included in our analyses between 2011 and 2023 (Table 1). These video samples were not missing any data and all 41 video frames could be read. Note that no videos were collected in 2020 due to the covid-19 pandemic and that only approximately 70% of the videos collected in 2021 and 2022 could be read due to more videos being collected in those years compared to previous years and the lack of additional video readers. The videos read by readers in those years were randomly selected. The seasonality, depth, latitude, and bottom water temperature of sampling was highly consistent among years (Table 1). Likewise, the spatial distribution of sampling was similar across years (Figure 1). Species-specific indices of abundance follow in alphabetical order by scientific name.

Table 1. Annual sampling information for samples included in these analyses. Minimum and maximum values are shown for date, depth, latitude, and bottom temperature. No sampling occurred in 2020 due to the covid-19 pandemic.

Year	*N*	Date	Depth (m)	Latitude (ºN)	Bottom temperature (ºC)
2011	580	5/19–10/26	15–93	27.2–34.5	14.8–28.8
2012	1083	4/24–10/10	15–106	27.2–35.0	12.9–27.8
2013	1221	4/24–10/4	15–100	27.3–35.0	12.4–28.1
2014	1382	4/23–10/21	15–110	27.2–35.0	16.1–29.3
2015	1406	4/21–10/22	16–110	27.3–35.0	13.6–28.5
2016	1410	5/4–10/26	17–115	27.2–35.0	15.5–29.3
2017	1422	4/26–9/29	15–111	27.2–35.0	14.8–28.2
2018	1653	4/25–10/4	16–114	27.2–35.0	13.6–28.3
2019	1544	4/30–9/25	16–110	27.2–35.0	15.0–29.5
2020	0	-	-	-	-
2021	1381	4/28–9/30	16–109	27.2–35.0	15.1–28.4
2022	1060	4/26–9/27	17–113	27.2–35.0	14.6–32.5
2023	1355	5/16–10/11	15–121	27.2–35.0	15.8–28.3

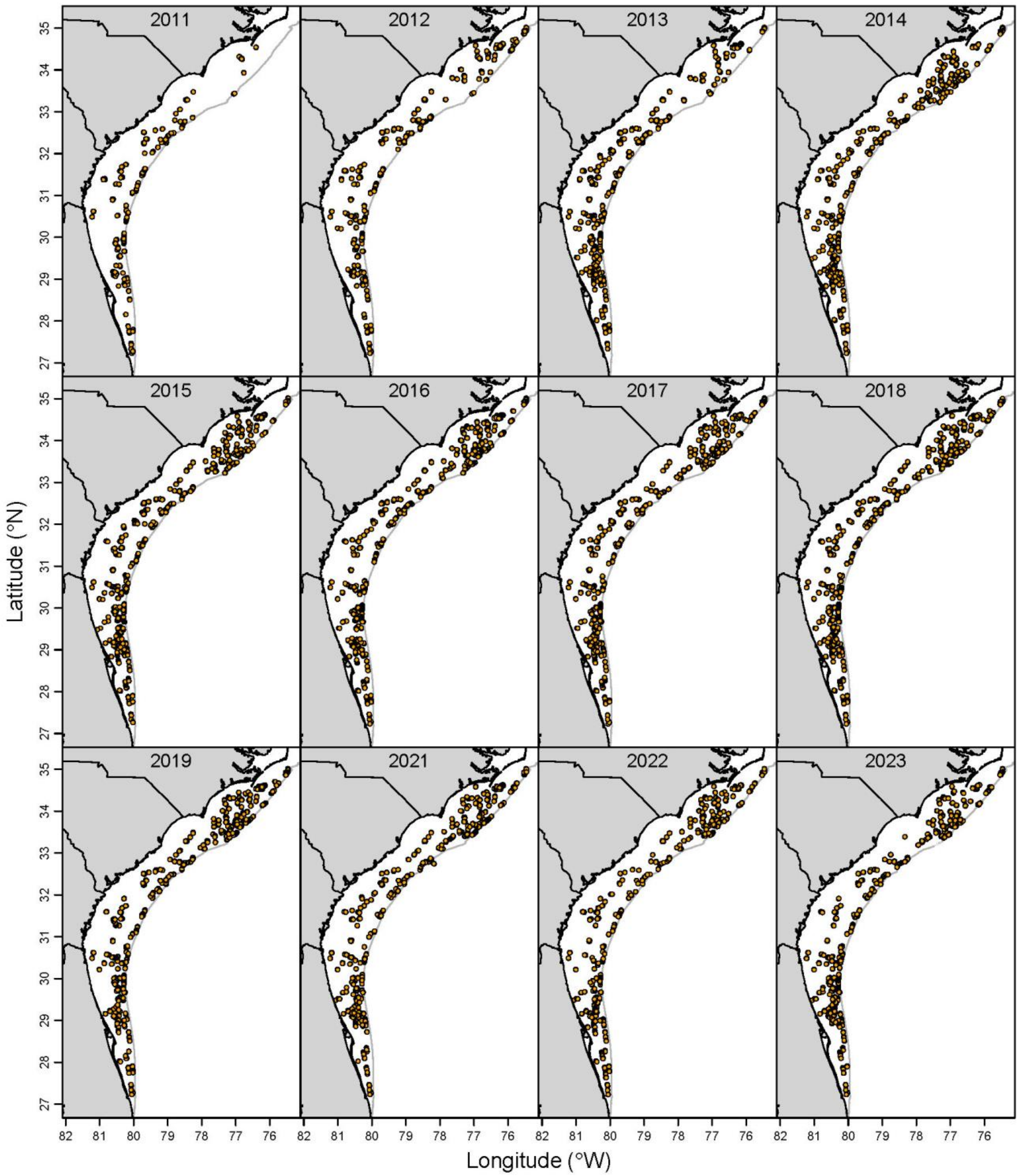
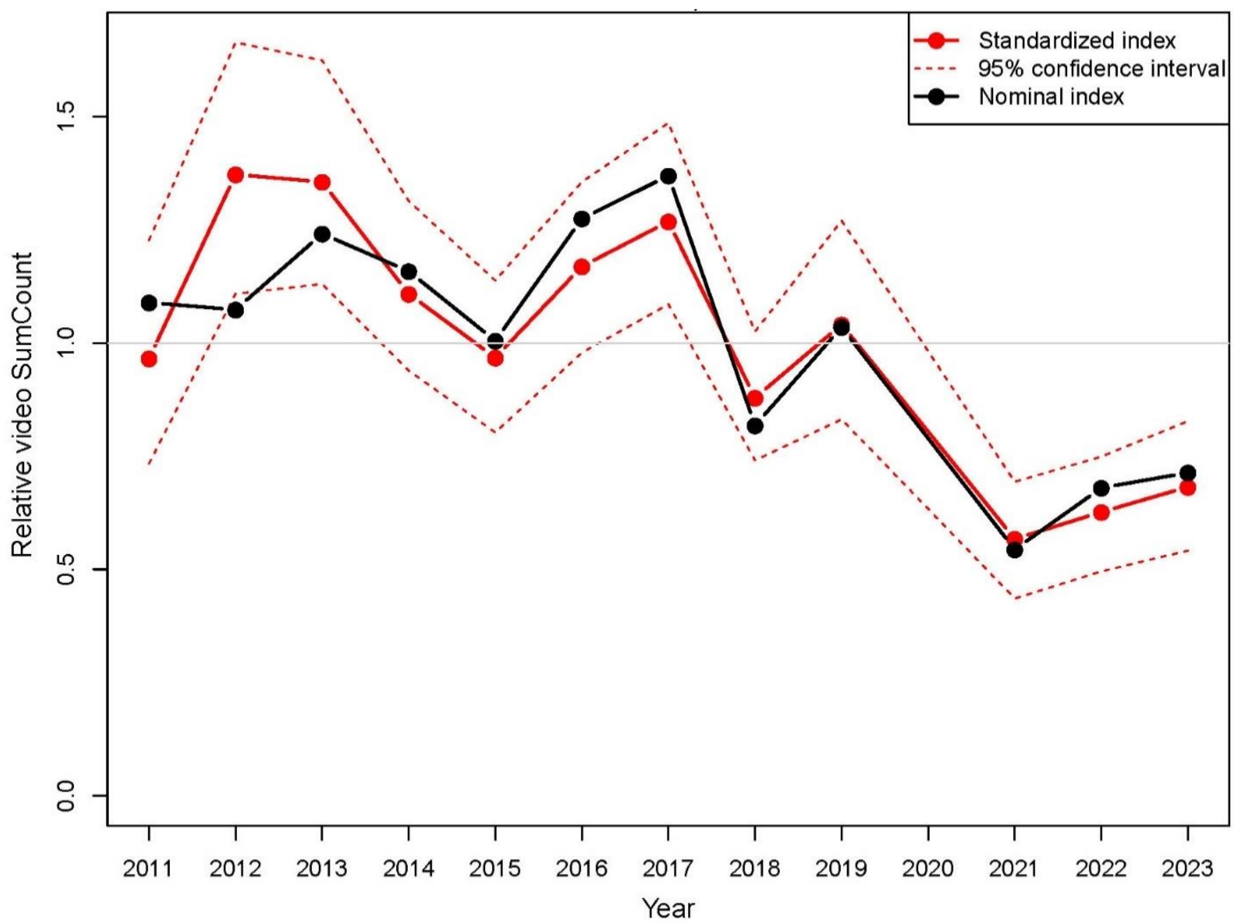


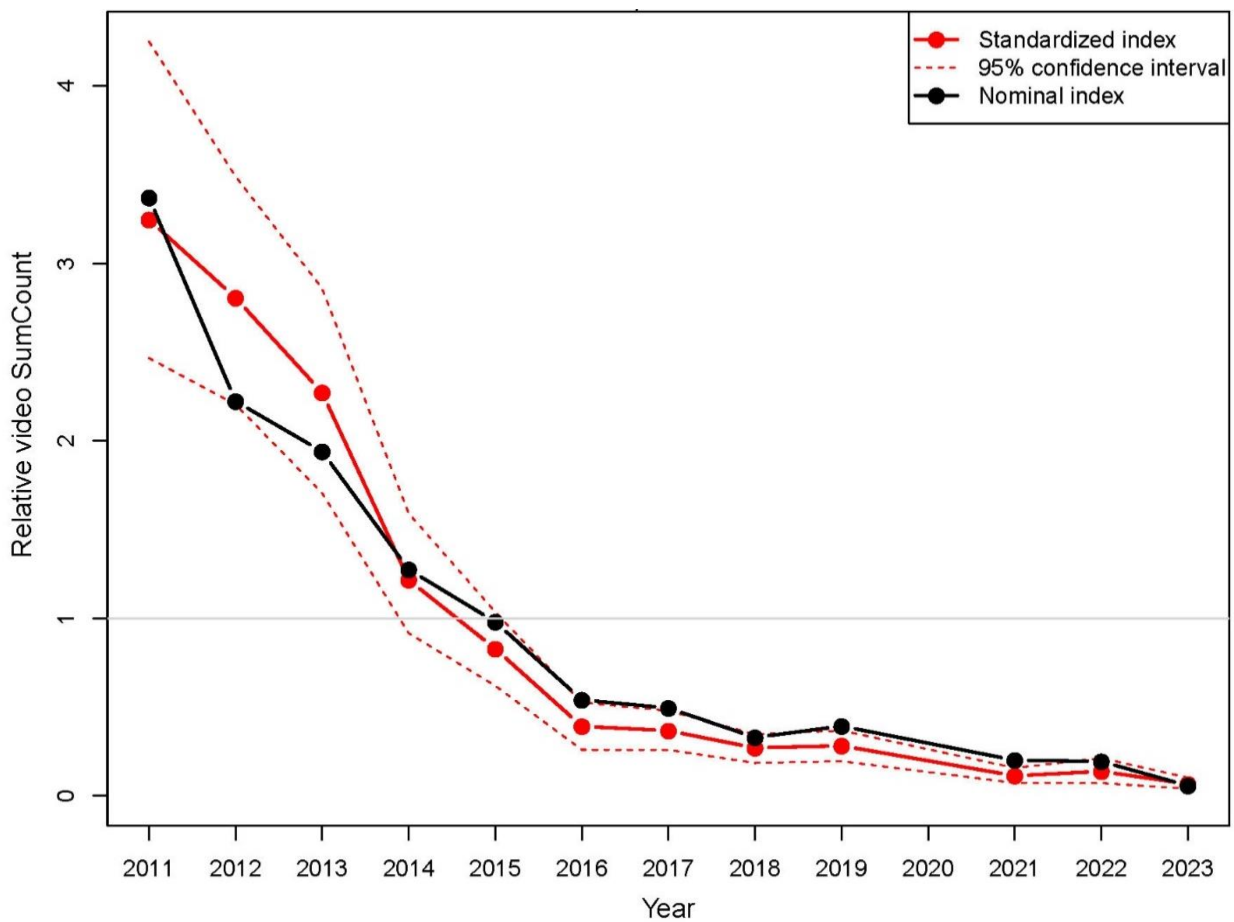
Figure 1. Annual sampling locations by the Southeast Reef Fish Survey that were included in the video analyses. Note that points often overlap and that no sampling was carried out in 2020 due to the covid-19 pandemic.

Gray triggerfish (*Balistes capriscus*)



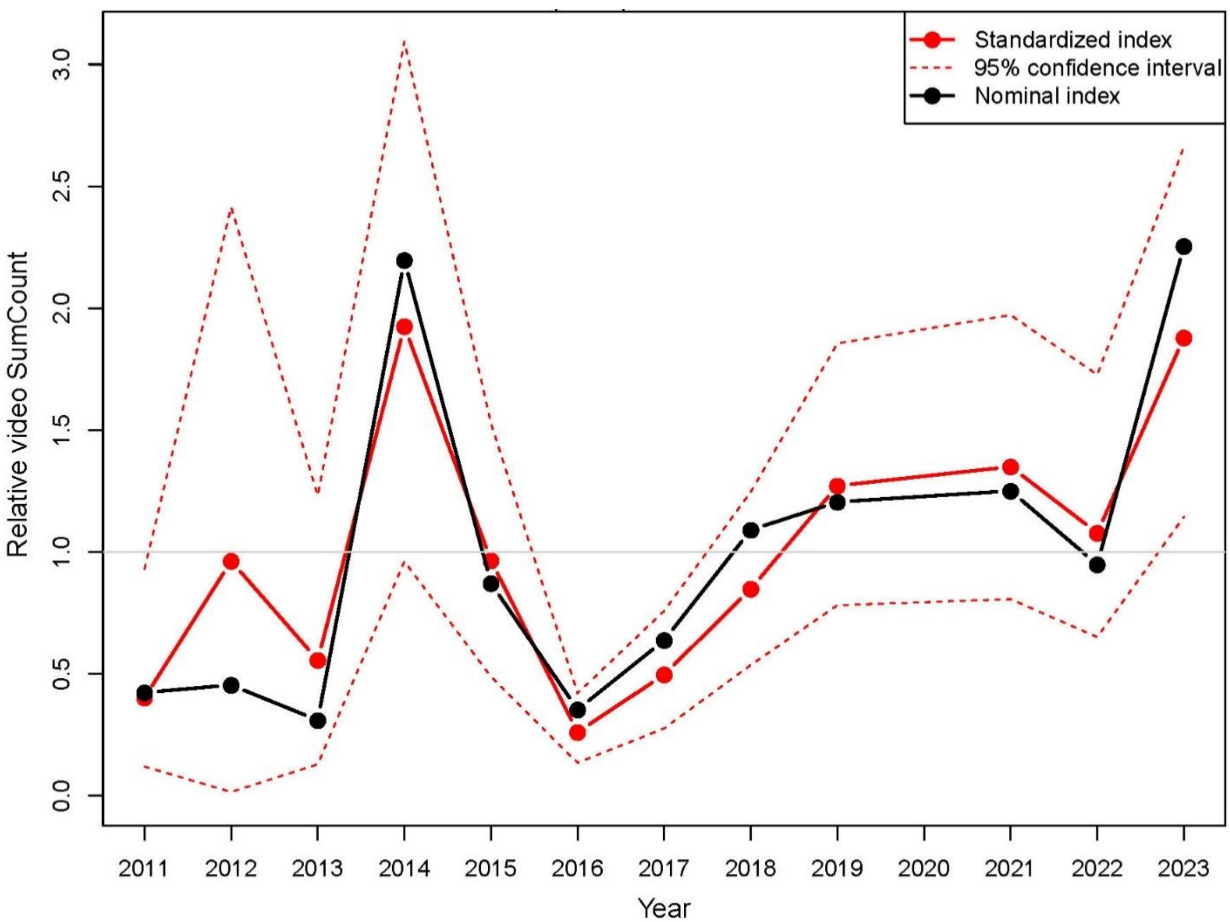
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.314	1.089	0.966	0.14
2012	1083	0.297	1.073	1.372	0.10
2013	1221	0.307	1.241	1.355	0.09
2014	1382	0.344	1.158	1.108	0.09
2015	1406	0.373	1.005	0.967	0.09
2016	1410	0.423	1.275	1.169	0.08
2017	1422	0.431	1.369	1.268	0.08
2018	1653	0.349	0.818	0.879	0.08
2019	1544	0.341	1.035	1.041	0.11
2020	0	-	-	-	-
2021	1381	0.253	0.544	0.567	0.12
2022	1060	0.282	0.680	0.627	0.10
2023	1355	0.303	0.714	0.682	0.11

Black sea bass (*Centropristis striata*)



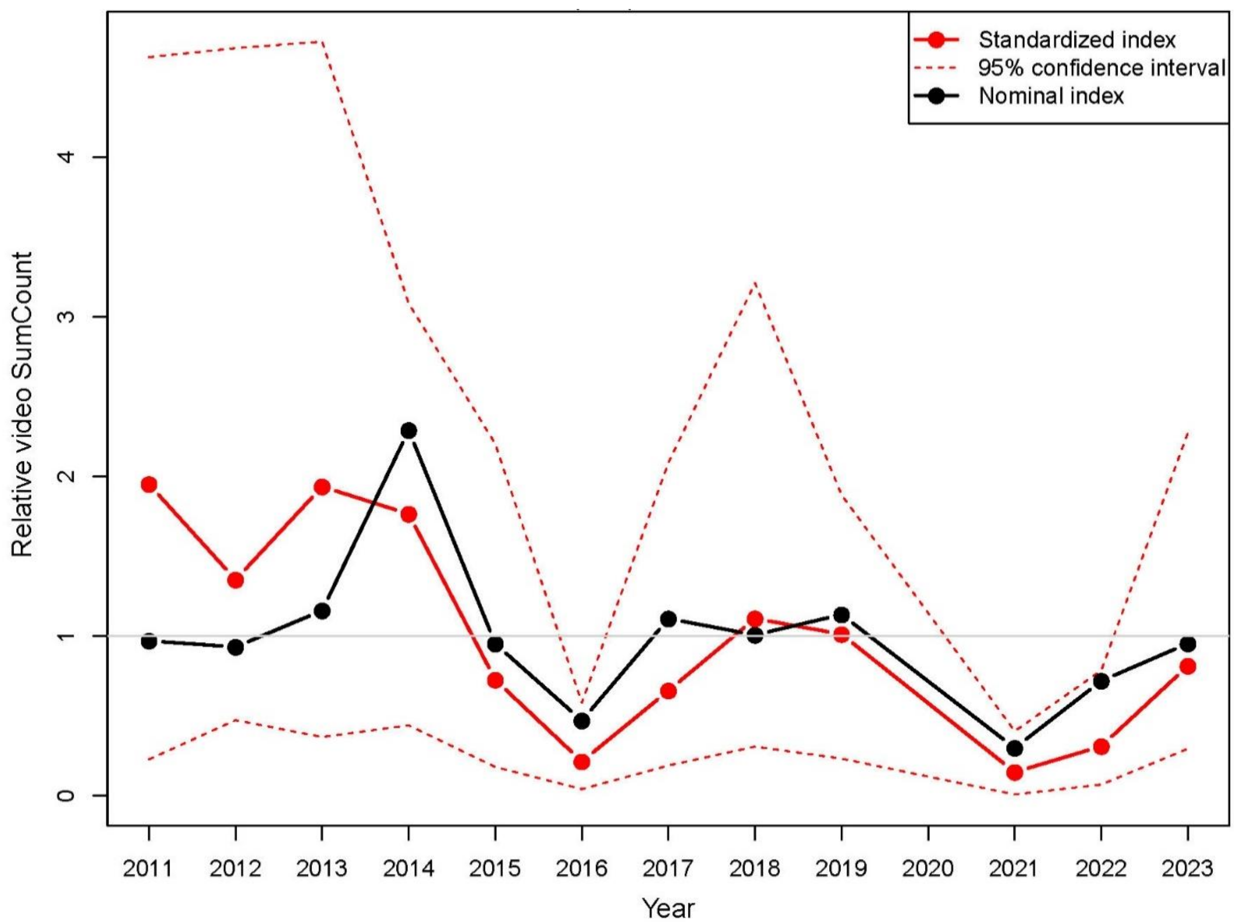
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.312	3.372	3.247	0.14
2012	1083	0.333	2.224	2.807	0.12
2013	1221	0.310	1.940	2.273	0.13
2014	1382	0.250	1.275	1.217	0.14
2015	1406	0.328	0.980	0.828	0.13
2016	1410	0.238	0.541	0.392	0.17
2017	1422	0.233	0.495	0.367	0.15
2018	1653	0.183	0.329	0.270	0.16
2019	1544	0.176	0.393	0.282	0.16
2020	0	-	-	-	-
2021	1381	0.105	0.201	0.113	0.19
2022	1060	0.123	0.194	0.138	0.27
2023	1355	0.070	0.057	0.067	0.24

Graysby (*Cephalopholis cruentata*)



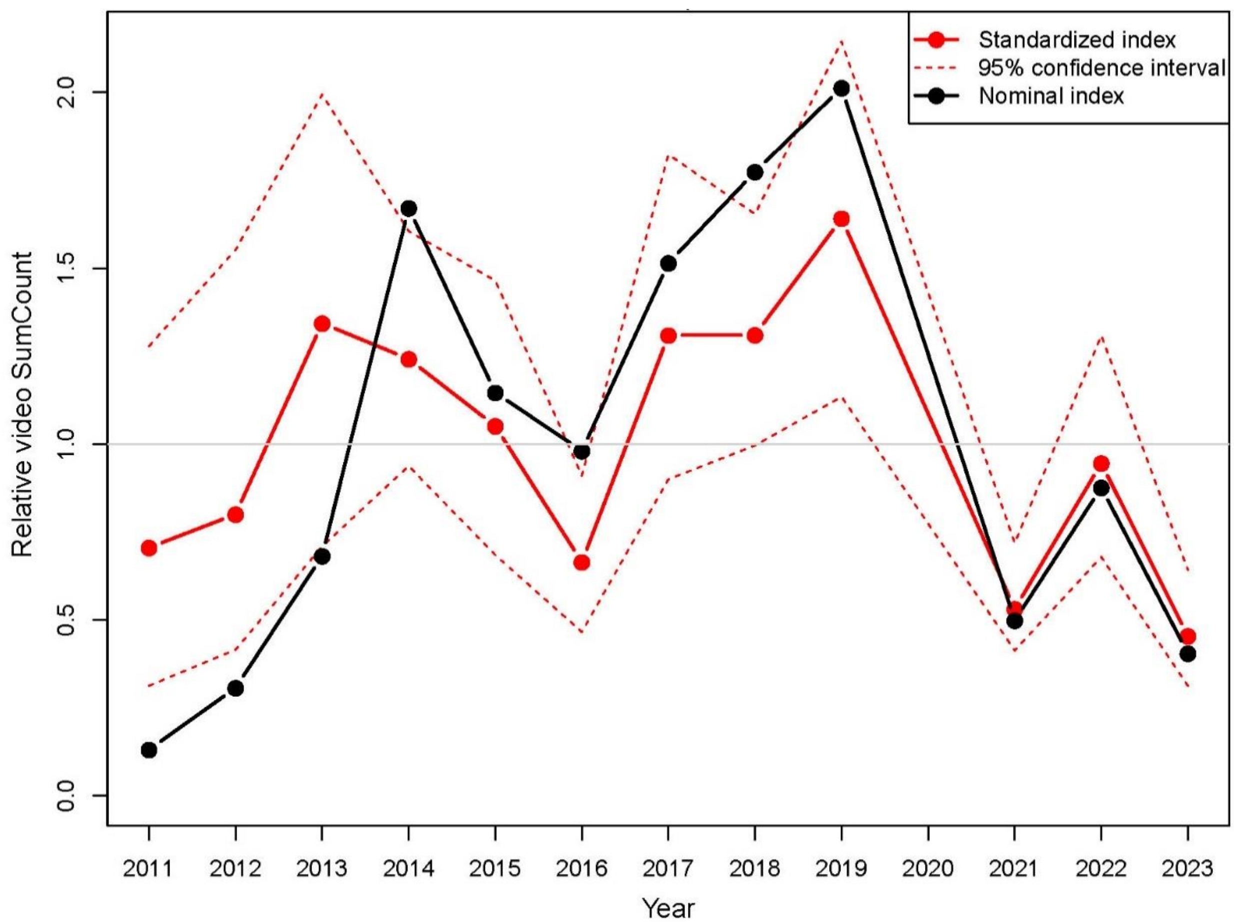
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.014	0.424	0.402	0.51
2012	1083	0.004	0.454	0.963	0.67
2013	1221	0.008	0.310	0.557	0.53
2014	1382	0.020	2.197	1.926	0.29
2015	1406	0.024	0.872	0.965	0.28
2016	1410	0.022	0.354	0.261	0.29
2017	1422	0.035	0.638	0.497	0.25
2018	1653	0.044	1.091	0.849	0.22
2019	1544	0.044	1.206	1.272	0.22
2020	0	-	-	-	-
2021	1381	0.042	1.251	1.350	0.22
2022	1060	0.041	0.949	1.078	0.24
2023	1355	0.054	2.255	1.879	0.22

Red grouper (*Epinephelus morio*)



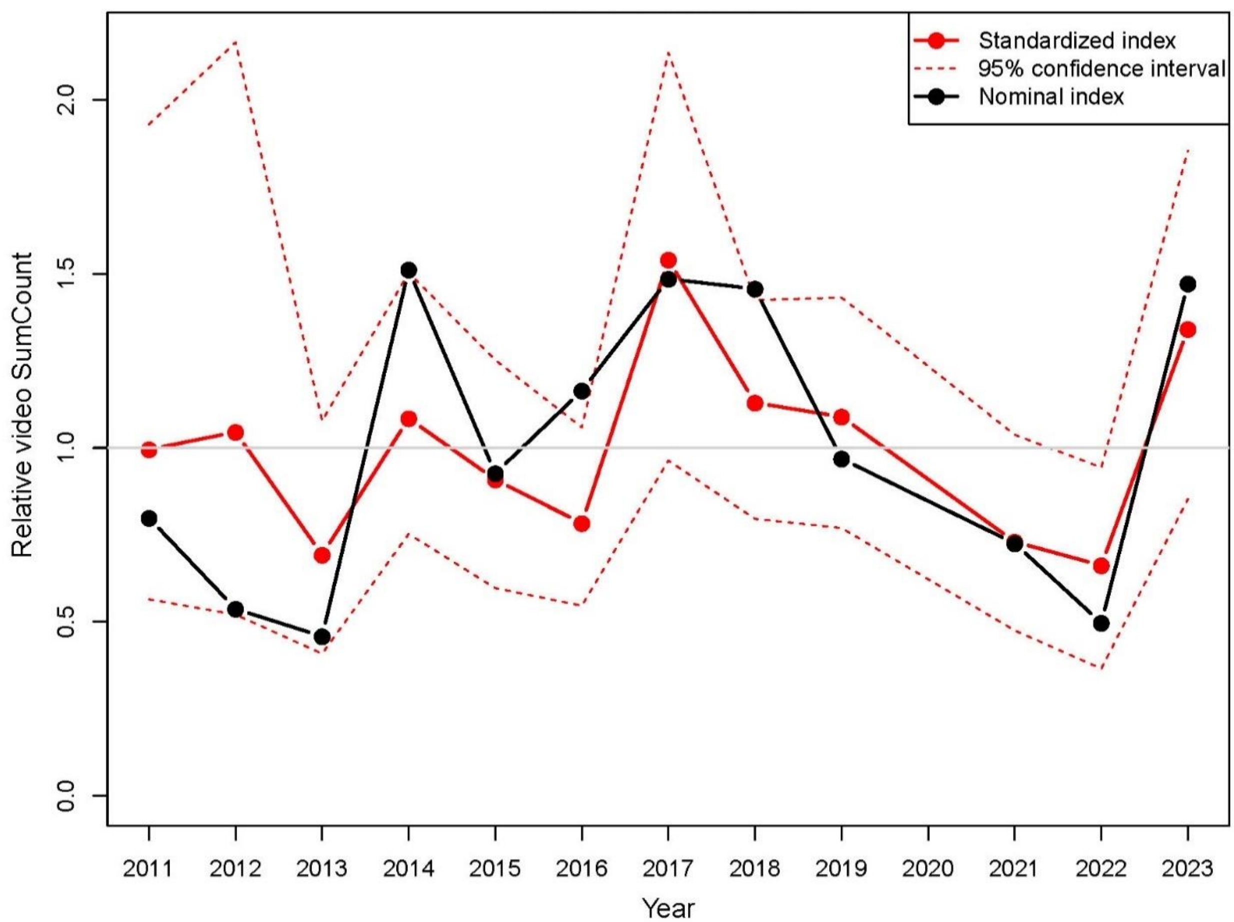
Year	N	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.014	0.970	1.951	0.67
2012	1083	0.018	0.932	1.353	0.65
2013	1221	0.010	1.160	1.936	0.62
2014	1382	0.021	2.289	1.765	0.50
2015	1406	0.016	0.952	0.726	0.65
2016	1410	0.010	0.471	0.215	0.87
2017	1422	0.012	1.110	0.659	0.78
2018	1653	0.016	1.007	1.110	0.60
2019	1544	0.020	1.135	1.012	0.54
2020	0	-	-	-	-
2021	1381	0.005	0.299	0.148	0.95
2022	1060	0.012	0.720	0.310	0.70
2023	1355	0.026	0.954	0.814	0.53

White grunt (*Haemulon plumierii*)



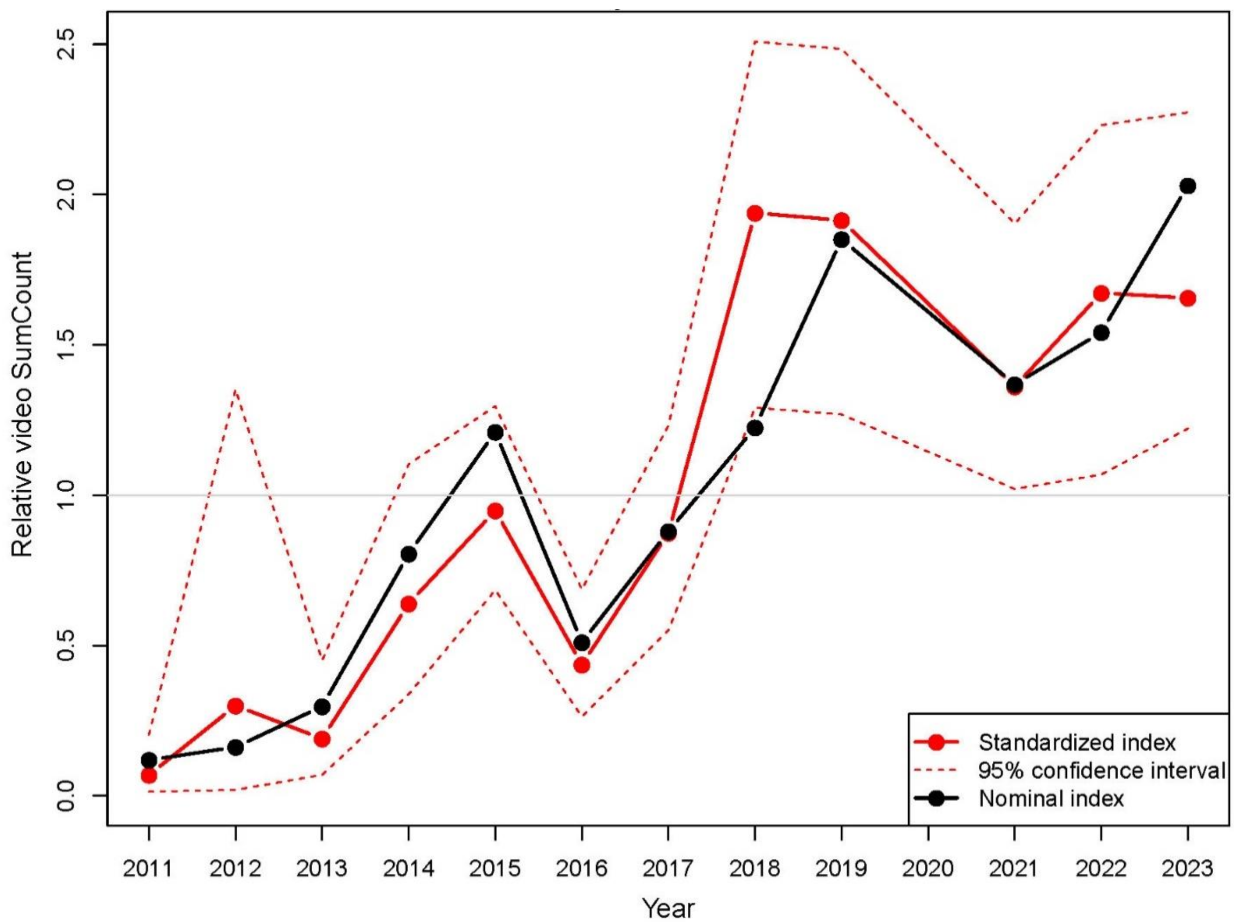
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.042	0.131	0.706	0.38
2012	1083	0.077	0.307	0.800	0.37
2013	1221	0.071	0.682	1.344	0.25
2014	1382	0.177	1.671	1.242	0.14
2015	1406	0.164	1.146	1.051	0.20
2016	1410	0.152	0.981	0.665	0.17
2017	1422	0.169	1.515	1.309	0.18
2018	1653	0.169	1.774	1.310	0.13
2019	1544	0.176	2.013	1.642	0.16
2020	0	-	-	-	-
2021	1381	0.138	0.499	0.531	0.14
2022	1060	0.137	0.876	0.946	0.17
2023	1355	0.114	0.405	0.454	0.18

Hogfish (***Lachnolaimus maximus***)



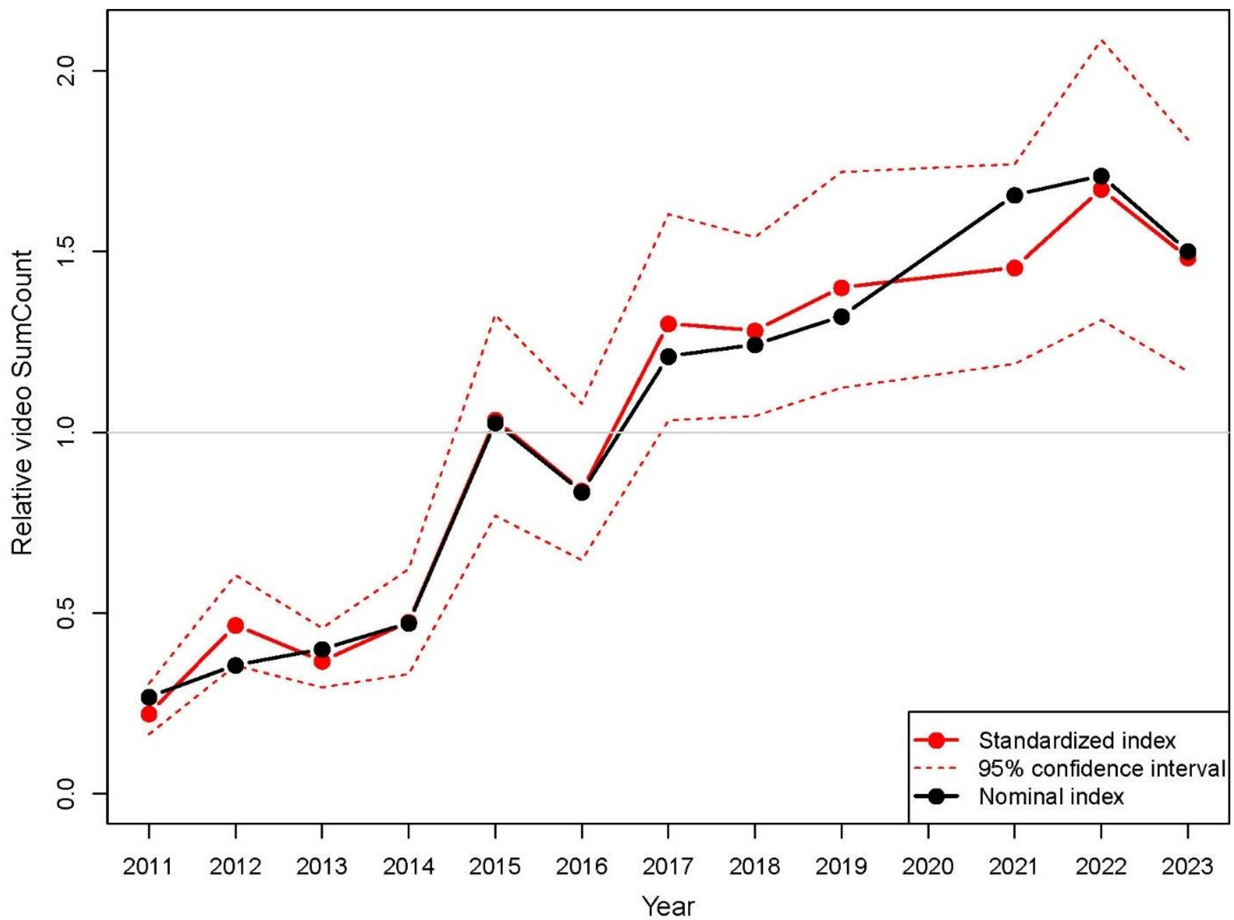
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.056	0.798	0.995	0.31
2012	1083	0.026	0.537	1.045	0.37
2013	1221	0.029	0.458	0.692	0.26
2014	1382	0.076	1.512	1.084	0.18
2015	1406	0.057	0.927	0.909	0.20
2016	1410	0.070	1.164	0.783	0.17
2017	1422	0.066	1.485	1.540	0.20
2018	1653	0.064	1.457	1.129	0.15
2019	1544	0.067	0.969	1.089	0.16
2020	0	-	-	-	-
2021	1381	0.054	0.725	0.730	0.19
2022	1060	0.037	0.497	0.662	0.24
2023	1355	0.069	1.472	1.341	0.19

Mutton snapper (*Lutjanus analis*)



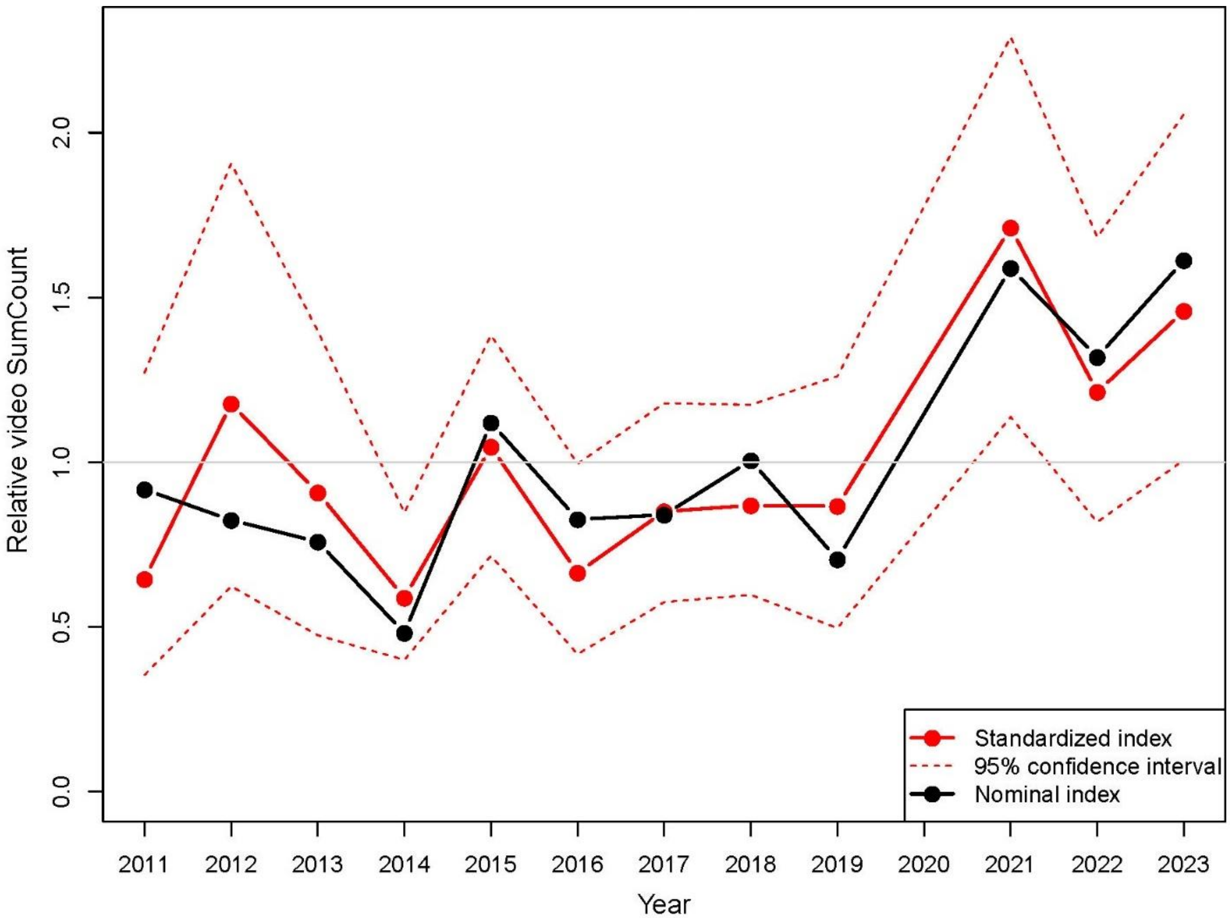
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.008	0.119	0.069	0.61
2012	1083	0.005	0.162	0.300	1.04
2013	1221	0.008	0.297	0.190	0.54
2014	1382	0.026	0.805	0.639	0.30
2015	1406	0.055	1.210	0.949	0.17
2016	1410	0.026	0.510	0.436	0.24
2017	1422	0.044	0.880	0.874	0.21
2018	1653	0.060	1.225	1.939	0.17
2019	1544	0.069	1.851	1.914	0.17
2020	0	-	-	-	-
2021	1381	0.075	1.368	1.361	0.16
2022	1060	0.068	1.542	1.672	0.19
2023	1355	0.088	2.030	1.656	0.15

Red snapper (*Lutjanus campechanus*)



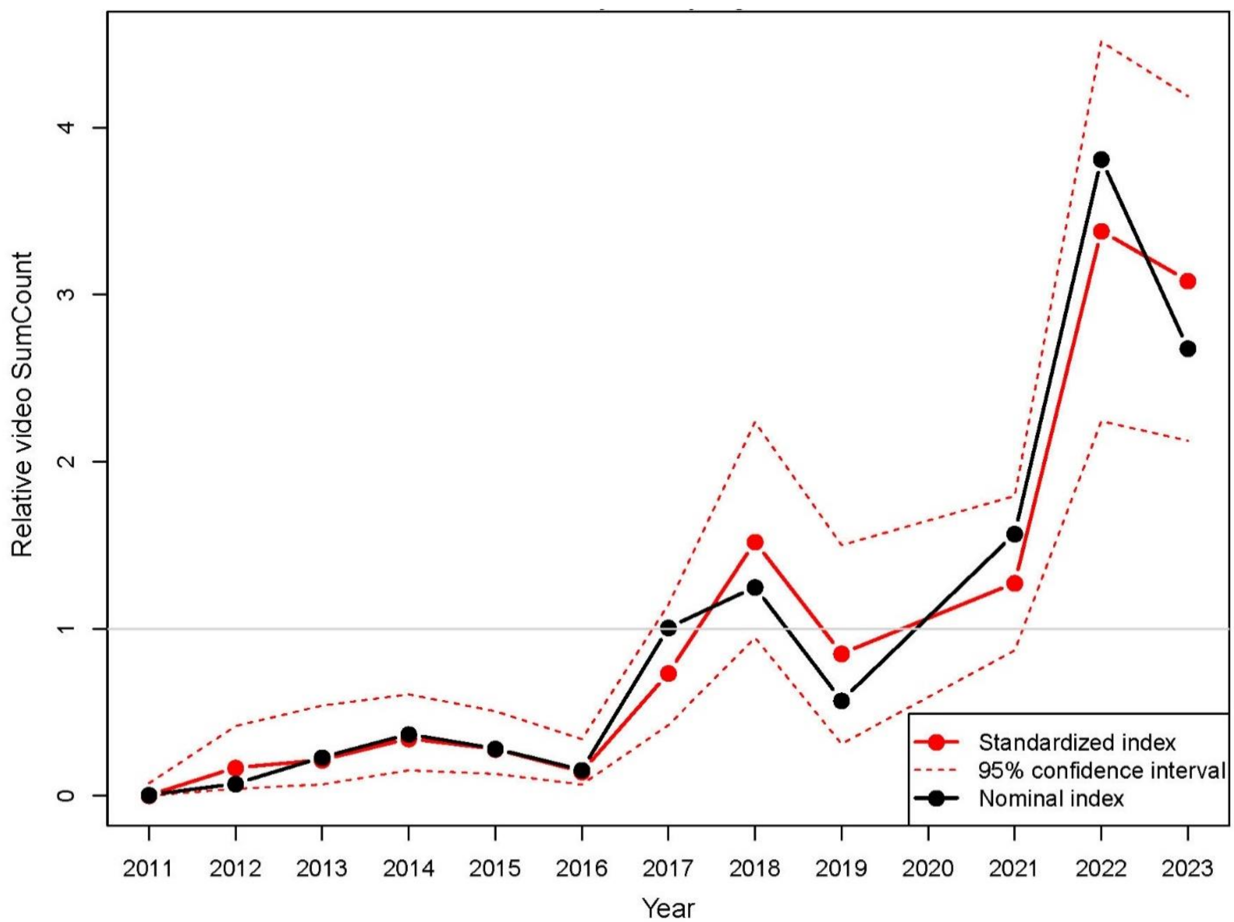
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.229	0.268	0.222	0.16
2012	1083	0.241	0.356	0.467	0.13
2013	1221	0.267	0.400	0.367	0.12
2014	1382	0.217	0.473	0.476	0.16
2015	1406	0.264	1.027	1.035	0.14
2016	1410	0.222	0.835	0.838	0.13
2017	1422	0.305	1.210	1.301	0.11
2018	1653	0.291	1.243	1.282	0.10
2019	1544	0.281	1.321	1.401	0.11
2020	0	-	-	-	-
2021	1381	0.358	1.657	1.456	0.10
2022	1060	0.339	1.710	1.673	0.12
2023	1355	0.334	1.501	1.483	0.10

Gray snapper (*Lutjanus griseus*)



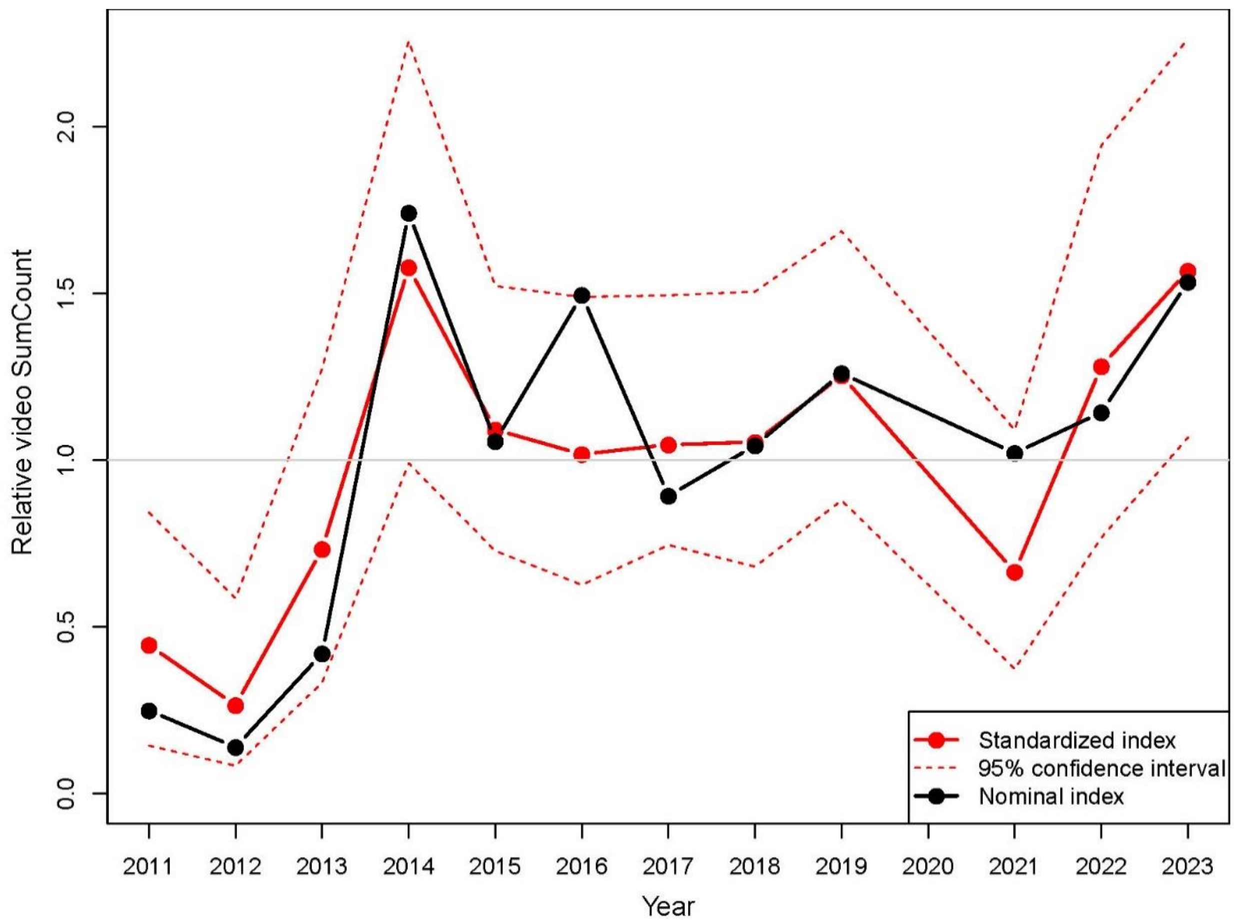
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.073	0.917	0.645	0.33
2012	1083	0.055	0.824	1.178	0.28
2013	1221	0.068	0.759	0.908	0.28
2014	1382	0.058	0.482	0.588	0.19
2015	1406	0.085	1.120	1.047	0.17
2016	1410	0.060	0.827	0.664	0.21
2017	1422	0.077	0.840	0.851	0.18
2018	1653	0.080	1.005	0.869	0.18
2019	1544	0.071	0.705	0.867	0.24
2020	0	-	-	-	-
2021	1381	0.096	1.589	1.712	0.17
2022	1060	0.114	1.319	1.213	0.18
2023	1355	0.115	1.613	1.459	0.18

Lane snapper (*Lutjanus synagris*)



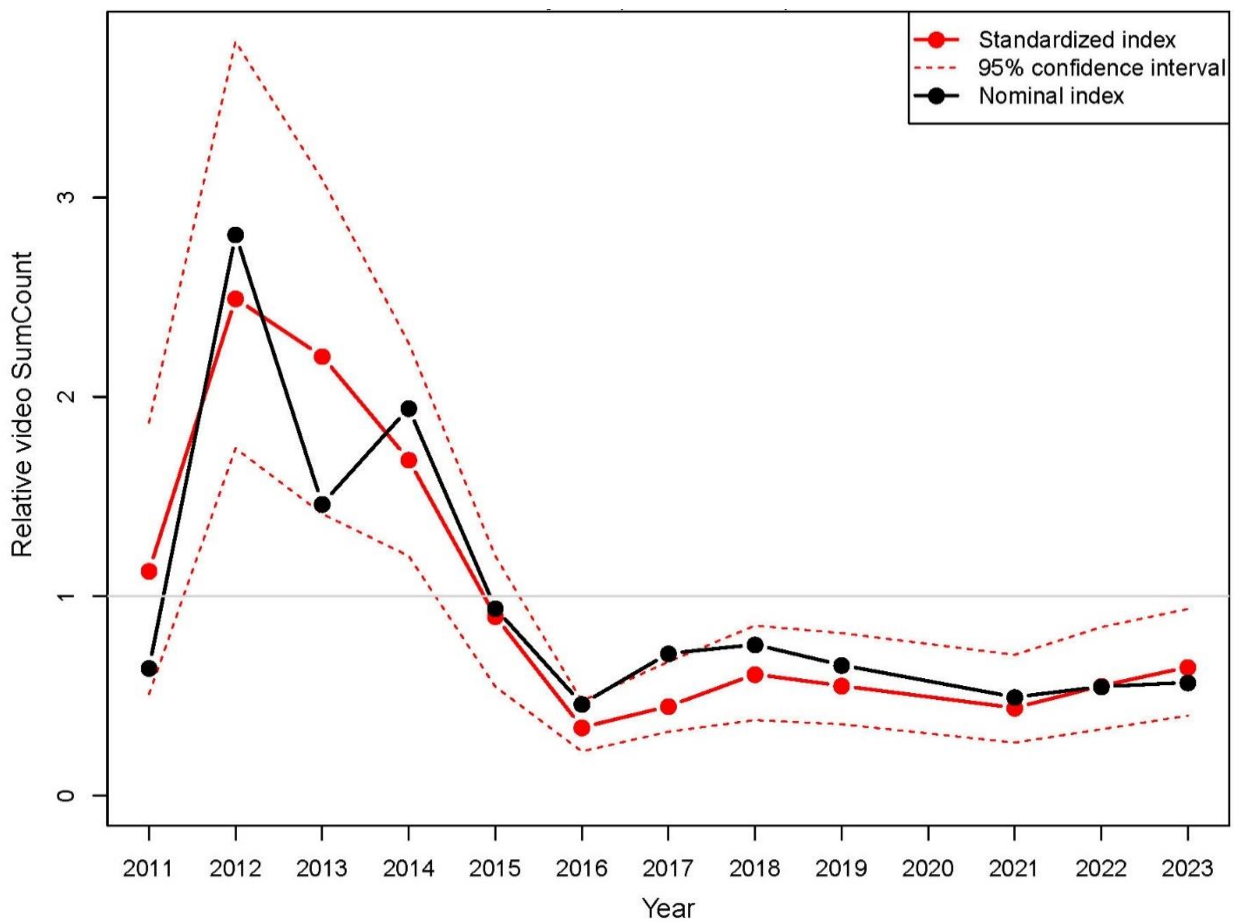
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.000	0.005	0.003	7.58
2012	1083	0.009	0.072	0.168	0.55
2013	1221	0.015	0.231	0.215	0.57
2014	1382	0.024	0.369	0.343	0.34
2015	1406	0.027	0.282	0.280	0.34
2016	1410	0.021	0.155	0.145	0.43
2017	1422	0.042	1.007	0.734	0.26
2018	1653	0.054	1.249	1.521	0.23
2019	1544	0.038	0.570	0.851	0.38
2020	0	-	-	-	-
2021	1381	0.074	1.569	1.274	0.19
2022	1060	0.098	3.811	3.382	0.17
2023	1355	0.075	2.679	3.084	0.18

Sand tilefish (*Malacanthus plumieri*)



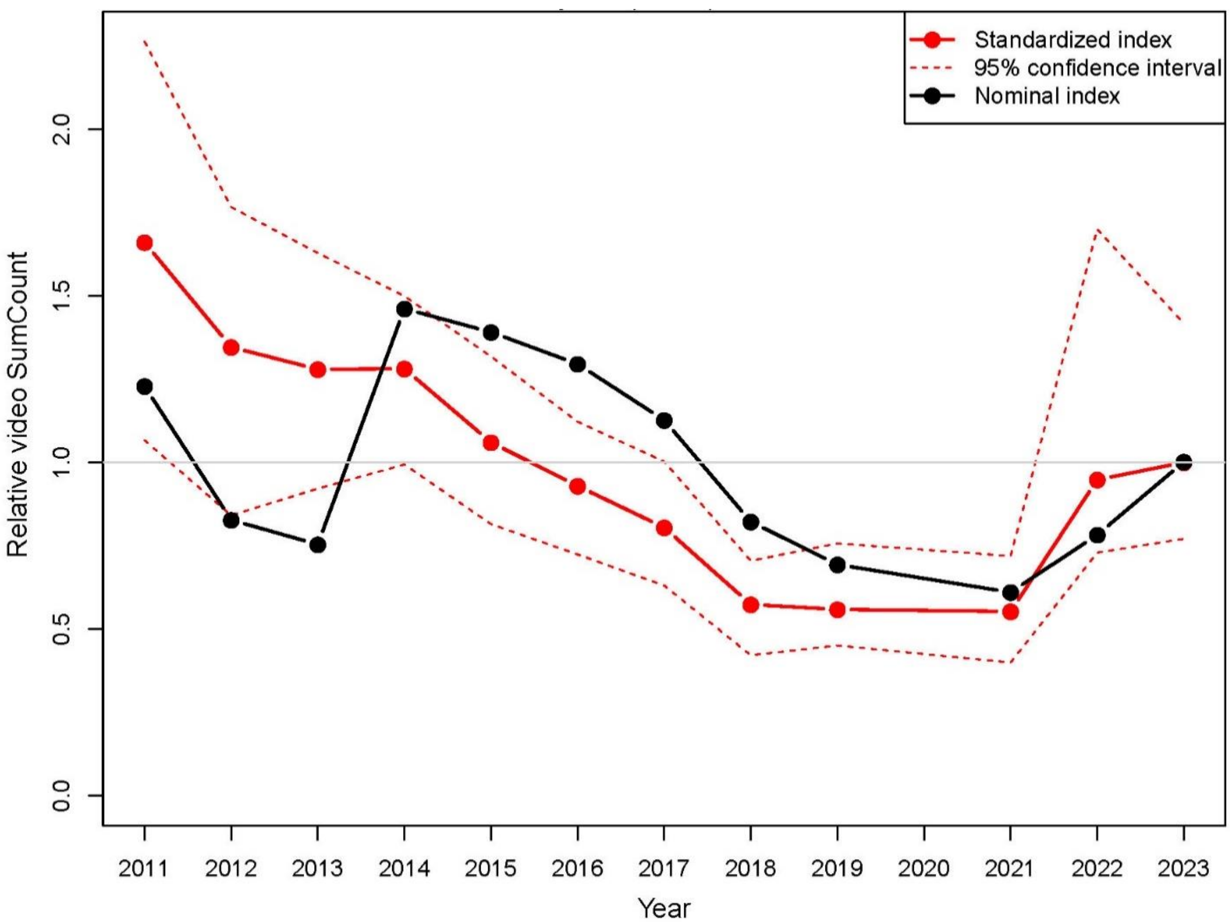
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.014	0.250	0.446	0.44
2012	1083	0.007	0.140	0.265	0.50
2013	1221	0.022	0.420	0.733	0.32
2014	1382	0.054	1.742	1.578	0.22
2015	1406	0.052	1.057	1.092	0.19
2016	1410	0.054	1.495	1.018	0.23
2017	1422	0.050	0.893	1.047	0.18
2018	1653	0.057	1.044	1.054	0.20
2019	1544	0.062	1.260	1.254	0.17
2020	0	-	-	-	-
2021	1381	0.037	1.021	0.665	0.28
2022	1060	0.043	1.143	1.281	0.23
2023	1355	0.058	1.535	1.568	0.18

Gag (*Mycteroperca microlepis*)



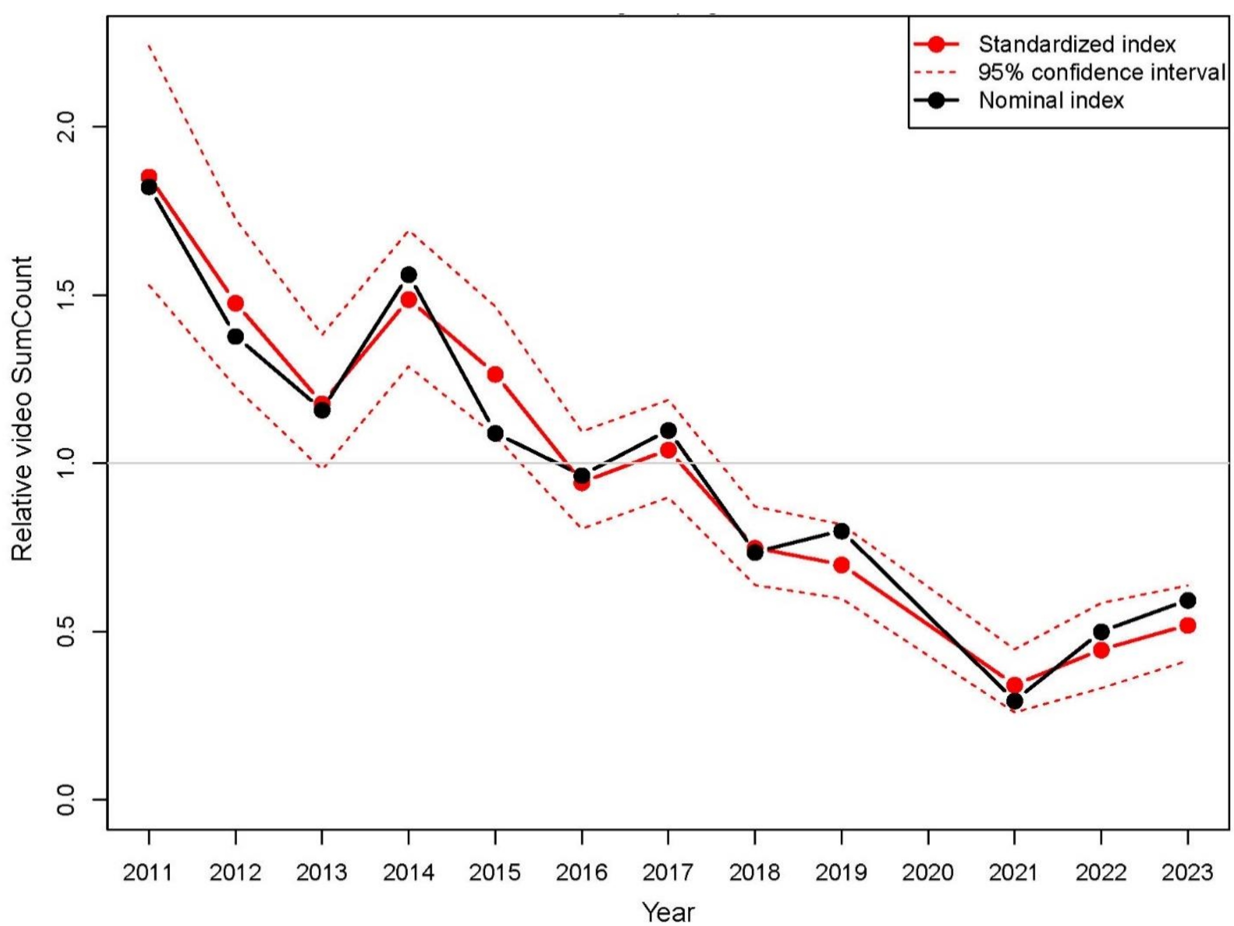
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.061	0.641	1.127	0.33
2012	1083	0.079	2.815	2.494	0.20
2013	1221	0.059	1.463	2.204	0.20
2014	1382	0.082	1.943	1.685	0.17
2015	1406	0.081	0.940	0.900	0.20
2016	1410	0.065	0.460	0.343	0.19
2017	1422	0.077	0.714	0.449	0.19
2018	1653	0.074	0.758	0.609	0.20
2019	1544	0.069	0.655	0.552	0.21
2020	0	-	-	-	-
2021	1381	0.057	0.495	0.441	0.26
2022	1060	0.065	0.548	0.551	0.24
2023	1355	0.063	0.568	0.645	0.21

Scamp (*Mycteroperca phenax*)



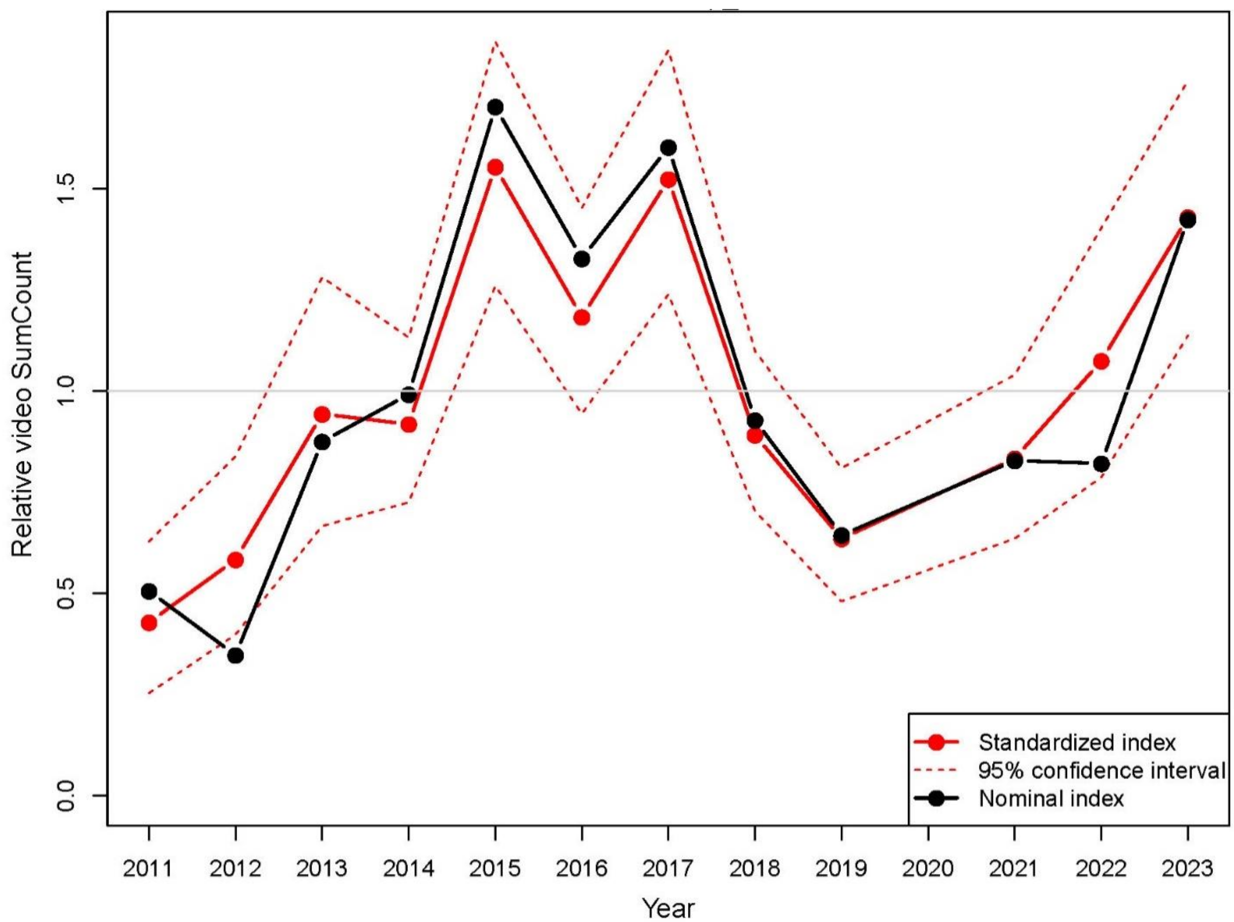
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.156	1.229	1.661	0.19
2012	1083	0.095	0.828	1.346	0.19
2013	1221	0.091	0.754	1.280	0.14
2014	1382	0.152	1.462	1.282	0.11
2015	1406	0.131	1.391	1.061	0.12
2016	1410	0.147	1.296	0.930	0.11
2017	1422	0.123	1.127	0.805	0.12
2018	1653	0.106	0.823	0.574	0.13
2019	1544	0.106	0.694	0.559	0.13
2020	0	-	-	-	-
2021	1381	0.083	0.611	0.553	0.15
2022	1060	0.097	0.783	0.949	0.23
2023	1355	0.106	1.002	1.000	0.16

Red porgy (*Pagrus pagrus*)



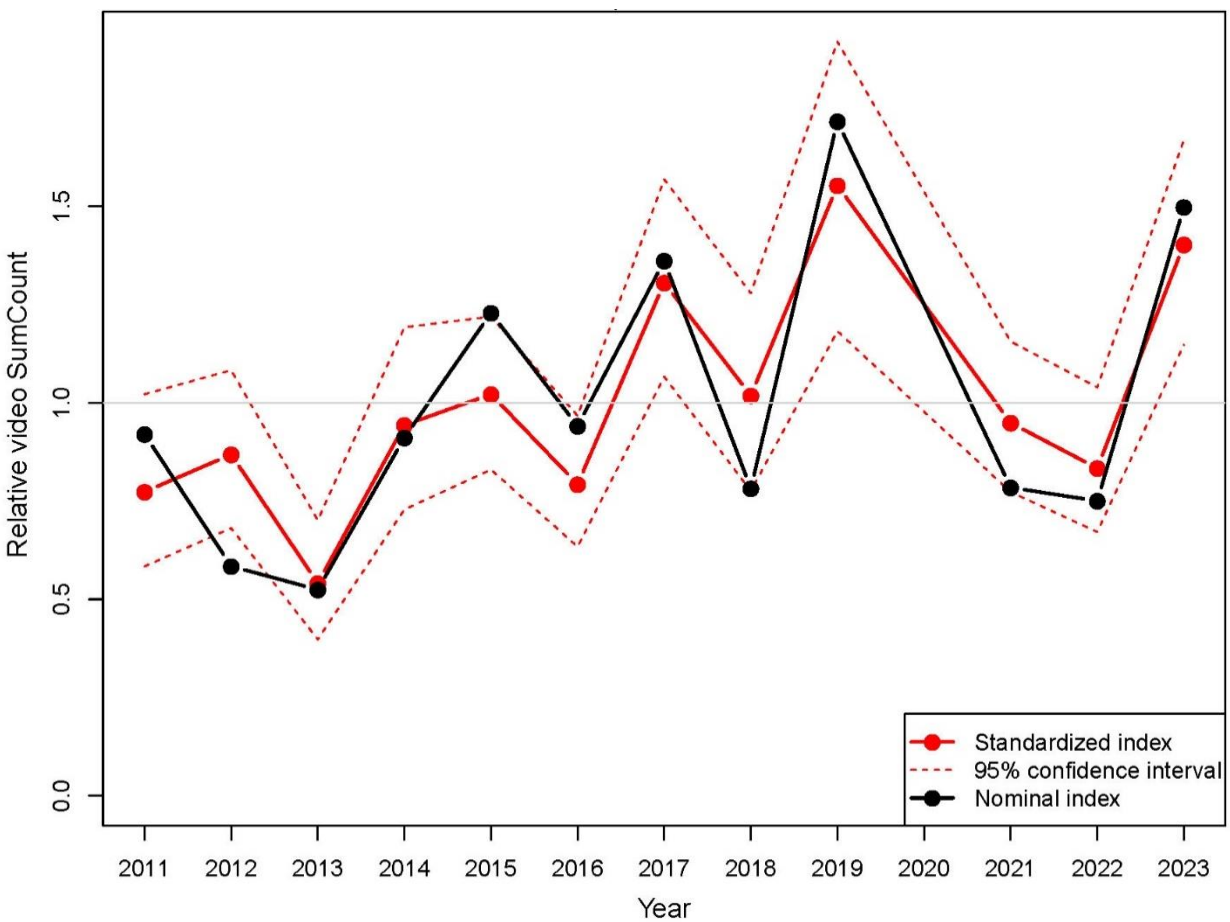
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.476	1.823	1.852	0.10
2012	1083	0.391	1.378	1.477	0.09
2013	1221	0.359	1.159	1.178	0.09
2014	1382	0.439	1.562	1.488	0.07
2015	1406	0.387	1.090	1.266	0.08
2016	1410	0.367	0.964	0.943	0.08
2017	1422	0.353	1.099	1.041	0.07
2018	1653	0.338	0.736	0.749	0.08
2019	1544	0.314	0.800	0.699	0.08
2020	0	-	-	-	-
2021	1381	0.145	0.295	0.342	0.14
2022	1060	0.175	0.500	0.446	0.14
2023	1355	0.241	0.594	0.520	0.11

Lionfish (*Pterois* spp.)



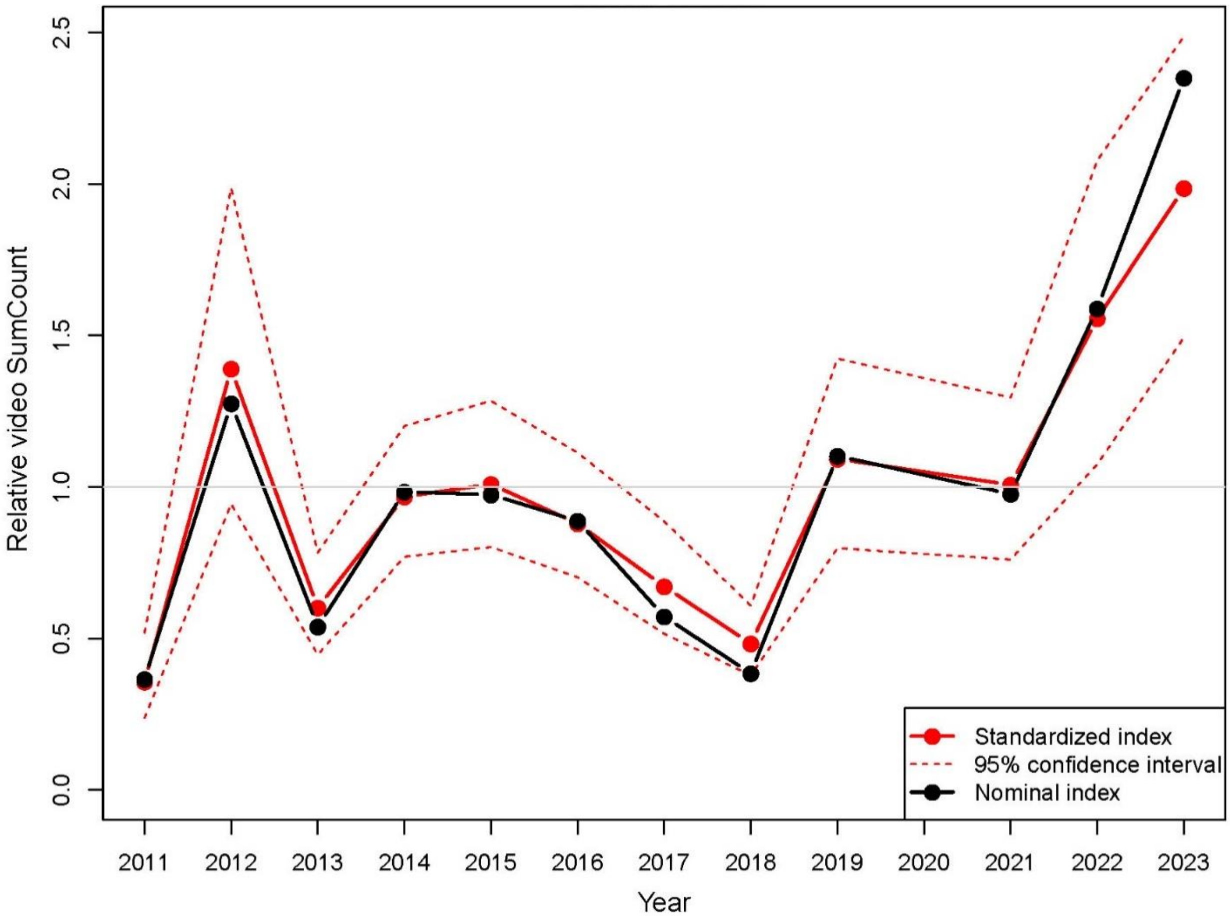
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.081	0.506	0.429	0.22
2012	1083	0.066	0.348	0.584	0.20
2013	1221	0.092	0.875	0.943	0.17
2014	1382	0.147	0.992	0.918	0.11
2015	1406	0.196	1.703	1.554	0.10
2016	1410	0.178	1.328	1.183	0.11
2017	1422	0.187	1.603	1.524	0.10
2018	1653	0.150	0.928	0.892	0.11
2019	1544	0.100	0.644	0.636	0.13
2020	0	-	-	-	-
2021	1381	0.108	0.829	0.833	0.13
2022	1060	0.126	0.821	1.075	0.15
2023	1355	0.161	1.424	1.430	0.11

Vermilion snapper (*Rhomboplites aurorubens*)



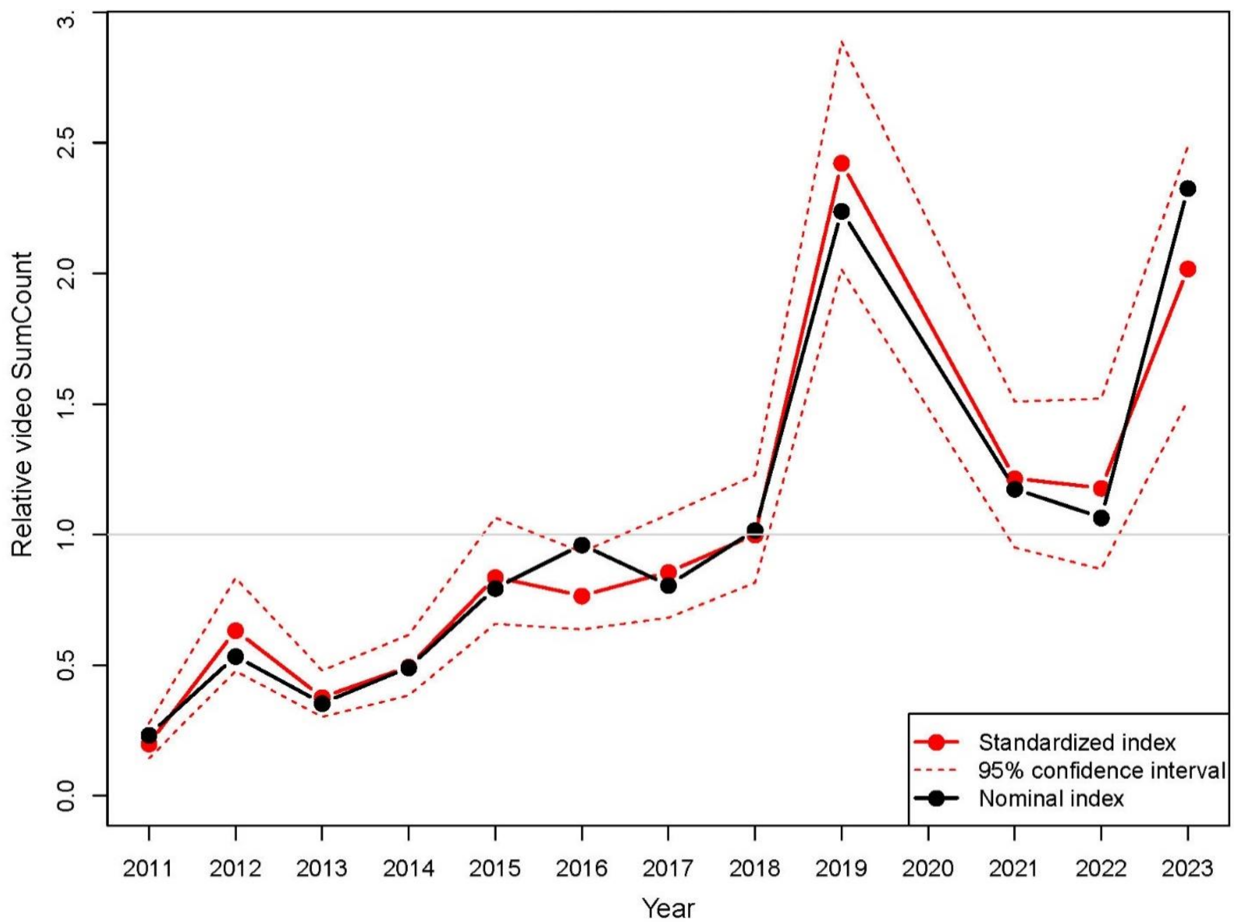
Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.322	0.920	0.773	0.14
2012	1083	0.253	0.584	0.868	0.12
2013	1221	0.241	0.524	0.541	0.15
2014	1382	0.305	0.911	0.943	0.12
2015	1406	0.312	1.229	1.021	0.10
2016	1410	0.332	0.941	0.793	0.10
2017	1422	0.327	1.361	1.305	0.10
2018	1653	0.310	0.782	1.018	0.13
2019	1544	0.316	1.716	1.553	0.13
2020	0	-	-	-	-
2021	1381	0.278	0.784	0.949	0.11
2022	1060	0.307	0.750	0.833	0.11
2023	1355	0.337	1.498	1.402	0.09

Greater amberjack (*Seriola dumerili*)



Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.110	0.365	0.357	0.20
2012	1083	0.155	1.275	1.390	0.20
2013	1221	0.143	0.539	0.601	0.14
2014	1382	0.203	0.984	0.968	0.11
2015	1406	0.227	0.974	1.009	0.12
2016	1410	0.225	0.888	0.879	0.12
2017	1422	0.165	0.572	0.672	0.14
2018	1653	0.142	0.385	0.483	0.12
2019	1544	0.198	1.102	1.093	0.14
2020	0	-	-	-	-
2021	1381	0.181	0.977	1.007	0.14
2022	1060	0.203	1.589	1.556	0.17
2023	1355	0.244	2.350	1.986	0.13

Almaco jack (*Seriola rivoliana*)



Year	*N*	Proportion positive	Nominal index	Standardized index	CV
2011	590	0.147	0.233	0.200	0.17
2012	1083	0.187	0.534	0.634	0.15
2013	1221	0.168	0.355	0.378	0.12
2014	1382	0.192	0.491	0.494	0.12
2015	1406	0.257	0.795	0.837	0.12
2016	1410	0.294	0.961	0.766	0.10
2017	1422	0.271	0.806	0.857	0.12
2018	1653	0.302	1.017	1.000	0.10
2019	1544	0.397	2.239	2.423	0.09
2020	0	-	-	-	-
2021	1381	0.272	1.175	1.215	0.12
2022	1060	0.290	1.065	1.179	0.14
2023	1355	0.380	2.326	2.018	0.13

Methods – video sampling

We used fishery-independent trap video data collected by the Southeast Reef Fish Survey (SERFS) for these analyses. SERFS consists of three fishery-independent programs that sample collaboratively in the SEUS using identical methodologies to sample reef fishes: (1) the Southeast Fishery-Independent Survey, (2) the Marine Resources Monitoring, Assessment, and Prediction program of the South Carolina Department of Natural Resources, and (3) the Southeast Area Monitoring and Assessment Program – South Atlantic. All programs are funded by the National Marine Fisheries Service and work together to sample reef fishes in the region. We used SERFS video data from 2011 to 2023 here, a time when video data were collected by SERFS in a consistent manner.

Stations were selected for sampling using a simple random sampling design. Out of approximately 4,300 stations on or near hardbottom, approximately 1,500 were selected for sampling each year. Most stations sampled in this study were randomly selected stations. Some stations were sampled even though they were not randomly selected for sampling in a given year, primarily to increase efficiency while on research cruises. A small number of new hardbottom stations were sampled each year based on information from fishermen, charts, sonar mapping, or historical data, and included in the analyses if hardbottom was detected. Sampling occurred during daylight hours each year between April and October on the RV *Savannah*, RV *Palmetto*, SRVx *Sand Tiger*, and NOAA Ship *Pisces*.

SERFS attached video cameras to chevron traps. Chevron traps used in our study were 1.7 m × 1.5 m × 0.6 m in size, with a total volume of 0.91 m^3. They were constructed from plastic-coated galvanized 2-mm diameter wire mesh (mesh size = 3.4 × 3.4 cm). Each trap was baited with 24 menhaden (*Brevoortia* spp.). In our study, chevron traps were deployed in groups of 6 or fewer, and each trap was separated from other simultaneously soaking traps by at least 200 m to provide independence among samples (Bacheler et al. 2018).

SERFS attached high-definition video cameras over the mouth and nose of each trap to provide additional data on the abundance and distribution of reef fish since 2011. In 2011–2014, SERFS attached Canon Vixia HF-S200 video cameras in Gates HF-S21 housings over the mouth of each trap deployed, facing away from the trap. In 2015, the survey replaced Canon cameras with GoPro Hero 3+ or 4 cameras (calibration study described below). Fish were only counted on cameras attached over the mouth of each trap. However, an additional camera (GoPro Hero 1, 3+, 4, or Nikon Coolpix S210/S220) was placed over the nose of the trap in order to quantify habitat information in the opposite direction (Bacheler et al., 2014). In 2023, GoPro Hero 3+ or 4 cameras were replaced by GoPro Hero 9 cameras, but settings, resolutions, and fields-of-view were set exactly the same as GoPro Hero 3+/4 cameras so calibration was not required. Videos were excluded from our analyses if they were too dark to identify fish, out of focus, corrupt, or if the corresponding traps were bouncing or moving.

Video-based relative abundance was calculated using a derivation of the MeanCount approach (Schobernd et al., 2014). The most common video reading metric is *MaxN* (Ellis and DeMartini, 1995), which is the maximum number of individuals of a given species observed in a single video frame. Schobernd et al. (2014) showed that *MaxN* was nonlinearly related to true abundance, however, and proposed using the MeanCount approach instead because it was proportionally related to true abundance. MeanCount is calculated as the mean number of individuals of a given species across a series of frames within a video. A potential downside of MeanCount is that the precision of MeanCount may be lower than for *MaxN* (Campbell et al., 2015, but see Schobernd et al., 2014). In our study, MeanCount was calculated as the mean number of individuals of each species across snapshots spaced 30 seconds apart beginning 10 minutes after the trap landed on the bottom (to allow time for the trap to settle) and lasting a total of 20 minutes. Thus, we read 41 frames from each video in our study. For these analyses, we used a derivation of MeanCount called SumCount, which was simply the sum of all individuals of a particular species observed across all video frames in our analysis. When the number of frames read is the same, as was the case in our study, MeanCount and SumCount are exactly linearly related. SumCount was used here instead of MeanCount because our particular modeling approach required count (instead of continuous) data.

A side-by-side camera calibration study took place in 2014 to develop species-specific camera calibration factors between Canon and GoPro cameras. Paired cameras were deployed on 54 traps side-by-side, facing away from the trap mouth, and the subsequent videos were read using the SumCount metric for a variety of species. For each species in which video-based indices of abundance were developed, we calculated species-specific camera calibration coefficients. Fish were more abundant on GoPro compared to Canon videos, so video counts from GoPro cameras in 2015–2023 were decremented using these calibration coefficients to make data from those cameras analogous with data collected by Canon cameras in 2011–2014.

Characteristics of the site and water were obtained for each station sampled in our study. We used the vessel's echosounder to estimate depth (m) and the ship's global positioning unit to estimate latitude. Bottom water temperature (ºC) for each group of simultaneously deployed traps was measured using a "conductivity-temperature-depth" cast. Habitat variables were visually estimated from each of the two cameras attached to traps in our study, one of which (substrate composition) was included in our analyses (see Bacheler et al. [2014] for more details). The percent of the visible substrate that was hard-bottom (hereafter referred to as "substrate composition") was estimated for each camera, and a mean value was calculated for each station sampled. Current direction was estimated as "away", "sideways", or "towards" based on the movement of visible particles in the water relative to the view field of the video camera over the trap mouth. Last, water clarity was classified as "low" if substrate could not be seen, "moderate" if substrate could be seen but not the horizon, and "high" if the horizon was visible in the distance. Video samples were excluded from our analyses if any variables were missing or unknown.

Methods – data analysis

The recommendation of the video index workshop (Bacheler and Carmichael 2014) was to apply a zero-inflated modeling approach to the development of fishery-independent video indices. Zero-inflated models are valuable tools for modeling distributions that do not fit standard error distributions due to excessive number of zeroes. These data distributions are often referred to as "zero-inflated" and are a common condition of count based ecological data. Zero inflation is considered a special case of over-dispersion that is not readily addressed using traditional transformation procedures (Hall 2000, Zeileis et al. 2008). Due to the high proportion of zero counts found in our data set, we used a zero-inflated model approach that accounted for the high occurrence of zero values, as well as the positive counts. The model does so by combining binomial and count processes (Zuur et al. 2009, Zeileis et al. 2008). We assume that there are two biological processes that require different statistical distributions – one for presence-absence and one for SumCount given their presence.

To standardize each species' video SumCounts, we used a zero-inflated negative binomial model formulation as:

SumCount = $y + wc + cd + sc + d + t + lat + temp \mid y + wc + cd + sc + d + t + lat + temp$

where y = year, wc = water clarity, cd = current direction, sc = substrate composition, d = depth, t = day of the year, lat = latitude, and $temp$ = bottom water temperature, and all predictor variables were categorical. In this formulation, variables to the left of the "|" apply to the count sub-model and variables to the right apply to the binomial sub-model. For all ZINB models, we retained full models for simplicity because model selection is time consuming. All data manipulation and analyses were conducted using R version 4.3 (R Core Team 2023). Modeling was executed using the zeroinfl function in the countreg package (Zeileis and Kleiber 2017).

Uncertainty in indices was computed using a bootstrap procedure with N = 1,000 replicates. In each replicate, a data set of the original size was created by drawing observations (rows) at random with replacement. This was done by year to maintain the same annual sample size as in the original data. For each species, the full model was fitted to each bootstrapped data set and uncertainty (CVs and 95% confidence intervals) were computed for the estimated standardized indices. Both the nominal (raw) index and the ZINB standardized index were normalized (scaled) to a mean of 1.

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