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Estimating the Riverine Abundance of Green Sturgeon Using a Dual-Frequency Identification Sonar

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Abstract

    To determine the total number of Green Sturgeon *Acipenser medirostris* present in the Rogue River, Oregon, we compared plot sampling using a dual-frequency identification sonar (DIDSON), a density-based estimation technique combining the number of individuals detected and the area sampled, to a concurrent mark–recapture estimate. Using the DIDSON-based method, we estimated the total abundance of Green Sturgeon to be 223 (95% confidence interval = 180–266). The mark–recapture method resulted in an estimate of 236 individuals (150–424). The noninvasive DIDSON transect estimates resulted in tighter confidence intervals and required fewer technician hours to collect the data than did the mark–recapture method (37 h versus 232 h, respectively). Precise estimates of the abundance and distribution of Green Sturgeon are important components to species recovery and management. Thus, this new technique has the potential to greatly improve population monitoring and is an excellent tool to identify occupied habitats.

Many terrestrial and aquatic genera are imperiled, and freshwater fish species are among the most at risk (Ricciardi and Rasmussen 1999). Sturgeons (family Acipenseridae) are considered some of the most at-risk freshwater species, and Billard and Lecointre (2001) listed overfishing, habitat degradation, and pollution as the primary causes. Currently, six sturgeon species in the United States are listed under the Endangered Species Act (Adams et al. 2007). The Green Sturgeon *Acipenser medirostris* is an anadromous species that spawns in three rivers along the West Coast of the United States. The species is composed of two populations, the Northern Distinct Population Segment, which spawns in the Rogue

and Klamath River systems, and the Southern Distinct Population Segment (SDPS), which spawns in the Sacramento River system (Adams et al. 2007; Seesholtz et al. 2015). The SDPS is listed as a threatened species by the U.S. Endangered Species Act (NMFS 2006). Currently, the size and demographic composition of Green Sturgeon populations are unknown.

Previously, no direct methods have been used to monitor the population size or total number of adult Green Sturgeon. In a status assessment, Adams et al. (2007) reviewed the available indices of Green Sturgeon abundance and found that inconsistent sampling and estimation methods led to biases that impaired the ability of the authors to assess population sizes. These indices resulted from the harvest of the Yurok tribe in the Klamath River, assessments of White Sturgeon *Acipenser transmontanus* by California Fish and Wildlife in San Francisco Bay, and entrainment within a major water diversion in the California Central Valley. Israel and May (2010) provided a novel application to estimate SDPS breeding population size using larvae sampled downstream of spawning sites. Unfortunately, the sampling from their study occurred in the upper portion of the known SDPS spawning range, omitting breeders in the lower reaches. All of these methods result in an incomplete estimate of the breeding population size. Thus, the evaluation of the status of these two populations requires a monitoring method applicable throughout the entire range of the species.

Studies on the distribution of Green Sturgeon have traditionally relied upon the capturing and handling of individuals. Early investigators of the spatial distribution of Green Sturgeon analyzed the returns of external tags recaptured by fisherman (Miller 1972) or the detection of eggs and larvae to infer habitat usage (Kohlhorst 1976). These types of studies can provide insights into the geographical distributions of entire populations but suffer from small sample sizes that reduce the precision of these estimates. This problem has been circumvented by implementing acoustic tags and a spatially diverse network of passive-tag-detecting hydrophones to understand spawning migrations (Erickson and Webb 2007; Heublein et al. 2008), estuarine distribution (Lindley et al. 2011), and habitat preferences in the nearshore and riverine environments (Mora et al. 2009; Huff et al. 2011). Information on habitat use in the open ocean has also been provided by pop-off satellite archival tag data and trawl bycatch information (Erickson and Hightower 2007). The addition of actively tracking individuals has greatly informed our understanding of fine-scale distribution and individual movement (Erickson et al. 2002; Benson et al. 2006; Thomas et al. 2014).

We present a rapid and noninvasive method to assess adult Green Sturgeon abundance during the spawning period using dual-frequency identification sonar (DIDSON; Sound Metrics, Bellevue, Washington). The application of DIDSON in fisheries research has varied from assessments of abundance and distribution (Becker et al. 2011) to evaluations of escapement (Holmes et al. 2006; Pipal et al. 2010) and sturgeon behavior

(Crossman et al. 2011). To introduce this method, we present a comparison of a DIDSON-based transect estimation technique with a mark–recapture estimation technique based on multiple gill-net sampling events. Our study had two objectives. First, we compared the accuracy and precision of the two abundance estimation methods. Second, we compared the number of technician hours required in the field to gather data for the two methods.

METHODS

The Rogue River is a major river along the West Coast of the United States, draining approximately 13,000 km^2 of southwestern Oregon. Our study was conducted in the lower 73 km of the river during October 2007 (Figure 1). The average river discharge during this period was 70 m^3/s as measured at the U.S. Geological Survey gauging station near Agness (Station 14372300). In the Rogue River, Green Sturgeon are able to access the lower 118 km up to Raine Falls and are generally found in reaches greater than 5 m deep (Erickson et al. 2002). Our depth surveys identified 65 reaches or "habitat units" satisfying this criterion. In these habitat units we performed presence–absence surveys using DIDSON and subsequent abundance estimates by performing DIDSON transects, as well as gill-net sampling when sturgeon were identified as present.

A DIDSON acoustic camera operates similarly to a medical ultrasound apparatus, emitting high-frequency sound and compiling the returns into an image in real time. This occurs several times per second; thus, the resulting data creates a movie-like image of ensonified objects. We were able to distinguish substrate types (sand, sand waves, cobble, and boulders), smaller fish in the water column, trees, and other objects. Sturgeon are large bottom-oriented fish and are easily
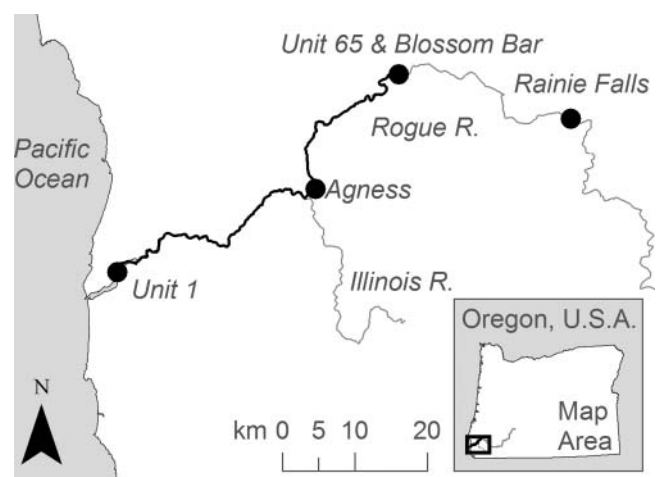

FIGURE 1. Map of the study area. The study reach is shown in black between units 1 and 65. Blossom Bar and unit 65 are in close proximity; thus, both are shown under the same black marker. The Rogue River upstream of the study area and the Illinois River are shown in gray.

differentiated from other fishes in the DIDSON record due to their large size, benthic orientation, and swimming style (Supplemental Video S.1 available in the online version of this article). Also, the DIDSON acoustic camera has a measure to determine the size of objects, thus calibrating the scale of what is viewed. By pausing playback of the files and measuring objects on screen, we were able to use size (approximately 2 m in length versus 1-m salmonids) as a criterion to identify sturgeon. Additionally, if the distance between the ensonified fish and its acoustic shadow is short (<1 m), the fish is bottom oriented. There are no other fish genera in the Rogue River that display these two characteristics. The DIDSON camera has two modes of operation, high frequency and low frequency. The high-frequency mode ensonifies a smaller area but images are clearer and show more detail than in the low-frequency mode. During the low-frequency mode, the DIDSON camera is able to view a larger area (approximately 15 m in width when water depth is near 7 m) but at a sacrifice to image clarity. We operated the DIDSON camera in low-frequency mode because the larger sampled area was preferable for detecting sturgeon presence and calculating their densities. We mounted the DIDSON camera to the gunwale of a jet boat using a custom-manufactured pan and tilt mount modified from Enzenhofer and Cronkite (2005).

Presence–absence surveys.—At each of the 65 habitat units, we sampled for the presence of sturgeon using DIDSON. At each unit, we performed a minimum of three transects with the DIDSON camera focused toward the bottom of the river, forward of the survey vessel. In this orientation the beam width was 29° oriented shore to shore and the beam height was 14° oriented top to bottom. Viewing window length was set to 20 m, and the window start was varied between 5 and 15 m depending on depth. During each transect, the survey vessel drove longitudinally through the entire length of the habitat unit, either upstream or downstream, while personnel viewed DIDSON images in real time. If sturgeon presence was confirmed, we estimated the number of sturgeon present using the two methods described below. If sturgeon were absent, we moved and surveyed the next unit.

Abundance estimation using DIDSON.—When sturgeon were present in a habitat unit, we used a plot-sampling abundance estimator to estimate the number of sturgeon present. We performed between three and seven transects and recorded unique DIDSON files for each transect. Transect paths were collected using a GEO XT GPS (Trimble, Sunnyvale, California). We viewed each DIDSON file three times and tallied the number of detected sturgeon in each file. When two counts were the same for a file, that number of detections was used as the number of detected sturgeon. The average of the three counts was used if the three counts disagreed. Transect widths were calculated from the DIDSON files using the measure tool in the DIDSON software as sampled width varies with depth

as well as DIDSON angle from horizontal. We measured the width of the DIDSON beam where it intersected the river bed at the 25th, 50th, and 75th percentile frames of each transect and calculated the mean to represent the transect width. The sampled area for each transect was calculated in ArcGIS (ESRI, Redlands, California) using a buffer around each transect path, with half the calculated transect width representing the buffer distance. The total sampled area per unit was calculated in ArcGIS as the minimum convex polygon containing the buffered transects.

We estimated the number of sturgeon present at each habitat unit and the total number of detected sturgeon using the following equations:

$$\hat{D}_i = \frac{\bar{y}}{\bar{a}}, \tag{1}$$

where $\hat{D}_i$ is the estimated sturgeon density at habitat unit i, $\bar{y}$ is mean number of sturgeon detected per transect, and $\bar{a}$ is the mean sampled area per transect. The total number of sturgeon at unit i was estimated as follows:

$$\hat{Y}_i = A_i \hat{D}_i, \tag{2}$$

where $\hat{Y}_i$ is the estimated number of sturgeon at unit i and A_i is the total sampled area at unit i. An estimated variance of the estimated mean density of sturgeon at unit i from transects j_{1-n} is calculated using the area-weighted least-squares variance estimator introduced here:

$$\hat{V}(\hat{D}_i) = \frac{\frac{1}{n}\sum \left(\frac{a_j}{\bar{a}}\right)^2 (\hat{D}_j - \hat{D}_i)^2}{n-1}. \tag{3}$$

An estimated variance of the estimated total number of sturgeon at unit i is

$$\hat{V}(\hat{Y}_i) = A_i^2 \hat{V}(\hat{D}_i). \tag{4}$$

An estimate of the total number of sturgeon detected during the sample period is

$$\hat{T} = \sum_i^n \hat{Y}_i. \tag{5}$$

An estimated variance of the estimated total number of sturgeon detected during the sample period is

$$\hat{V}(\hat{T}) = \sum_i^n \hat{V}(\hat{Y}_i). \tag{6}$$

Confidence intervals for the within-unit totals were calculated as

$$\mathrm{CI}_i = \hat{Y}_i \pm \sqrt{\hat{V}(\hat{Y}_i)}\, t_{(\alpha/2),\, n-1}, \tag{7}$$

where $t_{\alpha/2}$ is the entry in a one-sided t-distribution table for the desired α and n is the number of transects performed at habitat unit i. The 95% confidence intervals for the total number of detected sturgeon during the sample period were calculated as

$$\mathrm{CI}_{\mathrm{Total}} = \hat{T} \pm \sqrt{\hat{V}(\hat{T})}\, t_{(\alpha/2),\, n-1}. \tag{8}$$

White Sturgeon is present in our study area and indistinguishable from Green Sturgeon using DIDSON. We estimated the proportion of detected sturgeon that were Green Sturgeon (P) from our captured records as the ratio of captured Green Sturgeon (C_G) to the total number of captured sturgeon (N_C):

$$\hat{P} = \frac{C_G}{N_C}, \tag{9}$$

which can be approximated as a binomial proportion with mean $\hat{P}$ and variance:

$$\hat{V}(\hat{P}) = \frac{\hat{P}(1 - \hat{P})}{N_C}. \tag{10}$$

We estimated the total number of detected Green Sturgeon ($\hat{T}_G$) as

$$\hat{T}_G = \hat{P}\hat{T} \tag{11}$$

To estimate the variance of $\hat{T}_G$, we used a form of the Delta Method applicable to two independent random variables (Seber 1982):

$$V(\hat{T}_G) = [(\hat{P})^2 \cdot \hat{V}(\hat{T})] + [(\hat{T})^2 \cdot \hat{V}(\hat{P})] + [\hat{V}(\hat{P}) \cdot \hat{V}(\hat{T})]. \tag{12}$$

Confidence intervals for the total number of detected Green Sturgeon were calculated as

$$\mathrm{CI}_{\hat{T}_G} = \hat{T}_G \pm \sqrt{(\hat{T}_G)}\, z_{(\alpha/2)}. \tag{13}$$

The DIDSON-based estimation method makes five assumptions:

1. A closed population. No sturgeon emigrates or immigrates during the surveys.
2. Detection is 100%. If a sturgeon is in the view of the DIDSON, then it is detected and tallied.

3. The calculated densities are unbiased. Thus, measurements of transect area and number of sturgeon detected are unbiased.
4. All locations where sturgeon are present are surveyed. No aggregating sites are omitted from the survey.
5. All sturgeon are in the sampled units. No sturgeon are in transit between units during the survey.

To evaluate how sturgeon densities and the number of transects influence the bias and precision of this estimation method, we performed sampling simulations using the R package WiSP (Zucchini et al. 2007). At three uniform distributions of $N = 5$, 25, and 50, we simulated 100 site visits consisting of 20 randomly placed transects per site visit in a 125×300-m habitat unit. Transects ran parallel the entire length of the habitat unit, similar to the field transects. Transect widths were set at 10 m wide. For each site visit we calculated two metrics. First, we calculated the running estimate after each transect using equation (1). Second, we calculated the running coefficient of variation (CV) of the estimate of the total using the ratio of the square root of equation (4) to equation (2).

Mark–recapture estimate.—We also estimated the abundance of sturgeon using gill nets in order to verify the DIDSON-based estimates. We deployed two 3.0-m $\times$ 30.2-m stretch gill nets at habitat units where sturgeon were identified as present from DIDSON sampling. These nets were fished for 1 h each, with 30 min between settings, for a total of three sets per habitat unit per day. We sampled in each unit for 3 d, with 1 d rest between site visits, resulting in nine sets per habitat unit. Captured sturgeon were marked through the base of the dorsal fin with a loop-ended spaghetti tag inscribed with a unique five digit numerical ID, implanted with a PIT tag, and released to the habitat unit where captured.

We analyzed the resulting data with closed-population mark–recapture models that make the following assumptions (from Krebs [1998]):

1. A closed population. No individuals emigrate or immigrate during the surveys.
2. All animals have the same probability of capture in each sampling occasion.
3. Marking individuals does not affect their probability of recapture.
4. All marks are retained between sampling occasions.
5. All marks are detected if individuals are recaptured.

We estimated the total number of sturgeon in the study area and the number of sturgeon present at each unit using the "closed captures–full likelihood p and c" model (Otis et al. 1978) in Program MARK (White and Burnham 1999). This model estimates the probability of capture (p), the probability of recapture (c), and the number of individuals never caught (f_0). We evaluated four models representing constant p and c,

constant p and time-varying c, constant p = constant c, and time-varying p = time-varying c.

For the estimate of total abundance in the study area, capture data were pooled across sites into three sampling occasions (i.e., sampling occasion 1 represented the first three net sets at all of the habitat units, sampling occasion 2 represented the fourth through sixth net sets at all the habitat units, and sampling occasion 3 represented the seventh through ninth net sets at all the habitat units). For the habitat unit specific abundance estimates, we aggregated the detection histories using the same method as above but for that habitat unit only. Model selection was performed by choosing the model with the highest Akaike information criteria for small samples (AIC_c) weight reported in program MARK. The four models were evaluated to determine if violations of the assumptions were driving model results. For example, if marking animals reduced their probability of recapture (c) then we would expect the models with the probability of capture (p) not equal to the probability of recapture (c) to receive the greatest AIC_c. To make comparisons to the habitat-unit-specific DIDSON abundance estimates, we estimated habitat-unit-specific mark–recapture abundances in program MARK using the model with lowest AIC_c.

Comparison of field effort.—To compare the effort required to perform these two abundance estimation methods, we calculated the number of technician hours expended to gather the respective field data. These two estimations do not include travel time among habitat units or time spent postprocessing and analyzing the data. For the mark–recapture estimate, we assumed that three technicians were required: one to pilot the survey vessel and two to deploy and retrieve the nets. All three technicians would participate in sturgeon processing and release. We tallied the total time from our datasheets when nets were deployed and factored in an additional 10 min per captured sturgeon to remove them from the net, process them, and release them. To calculate the total amount of technician hours required to complete the DIDSON field surveys, we assumed that two technicians were required: one to pilot the survey vessel and one to operate the DIDSON. These two technicians were estimated to spend 15 min surveying locations where sturgeon were absent and 30 min where sturgeon were present. We were unable to calculate the amount of time required for this task directly from the datasheets or DIDSON files as time was not marked during the data collection and the DIDSON files from locations where sturgeon were absent were not archived.

RESULTS

Transect simulations indicated that the DIDSON-based sampling method and implemented estimators were unbiased at low, medium, and high densities (Figure 2). In Figure 2, note that the solid lines at the center of the boxes, indicating the mean estimates using the DIDSON, coincided with the light gray lines indicating the true number of sturgeon in the simulation. This was true for estimates of 5, 25, and 50 uniformly distributed Green Sturgeon. Furthermore, these simulations suggest that a determination of the number of sturgeon at a given location can be estimated from a feasible number of transects (Figure 3). For example, when N = 25 or 50, we would require approximately seven transects to reach an estimate with an average CV less than or equal to 0.25. However, the estimation method was less precise at low densities (N = 5).

We detected sturgeon at 9 of the 65 locations surveyed using the DIDSON. To minimize poaching, we report the habitat units by their unit number and omit any spatial information, such as latitude and longitude or river kilometer, due to the limited number of locations where Green Sturgeon were present and the fact that these locations are occupied by sturgeon year after year (E.A.M., unpublished data). However, habitat units are numbered moving upstream, with unit 1 being the closest to the river mouth and unit 65 being nearest to the upstream extent of sampling below Blossom Bar.

The abundance of Green Sturgeon was estimated using the DIDSON at all locations where sturgeon were detected. The number of transects performed at each habitat unit varied between three and seven. Using this method, we estimated the total abundance of Green Sturgeon was 223 individuals within the 95% confidence limits of 180 and 266 (Table 1). During this period, sturgeon appeared to congregate in shoals, ranging from very few (unit 44, N = 6) to many (unit 15, N = 70) individuals. We calculated that DIDSON transects required a total of 37 technician hours to perform.

At seven of the locations where Green Sturgeon were detected, we estimated their abundance using mark–recapture estimation. We performed a total of 81 net sets, for a total soak time of 63 h and 14 min. As a result of the limited 3-week sample period, we were unable to sample habitat units 35 and 53 using gill nets. We sampled habitat unit 24 with gill nets but were unable to capture sturgeon. No recaptures occurred at habitat units 1 or 44. Our net sets resulted in 85 sturgeon encounters, consisting of 77 individuals (76 Green Sturgeon and 1 White Sturgeon) and 9 recaptures. All recaptures occurred in the same unit as their first capture. The single White Sturgeon was not recaptured and was not included in the mark–recapture estimates. This component of the study required a total of 232 h of technician hours to complete.

Of the four mark–recapture models we implemented to estimate the total number of Green Sturgeon in our study area, the model representing time-varying p = time-varying c resulted in the highest AIC_c weight of 0.979 (Table 2). We estimated that the number of Green Sturgeon in our study area was 236 within the 95% confidence limits of 150 and 424 (Table 3).

The DIDSON-based estimates of abundance agreed with the mark–recapture estimates and generally resulted in tighter confidence intervals. The habitat-unit-specific DIDSON abundance estimates and their 95% confidence intervals are almost
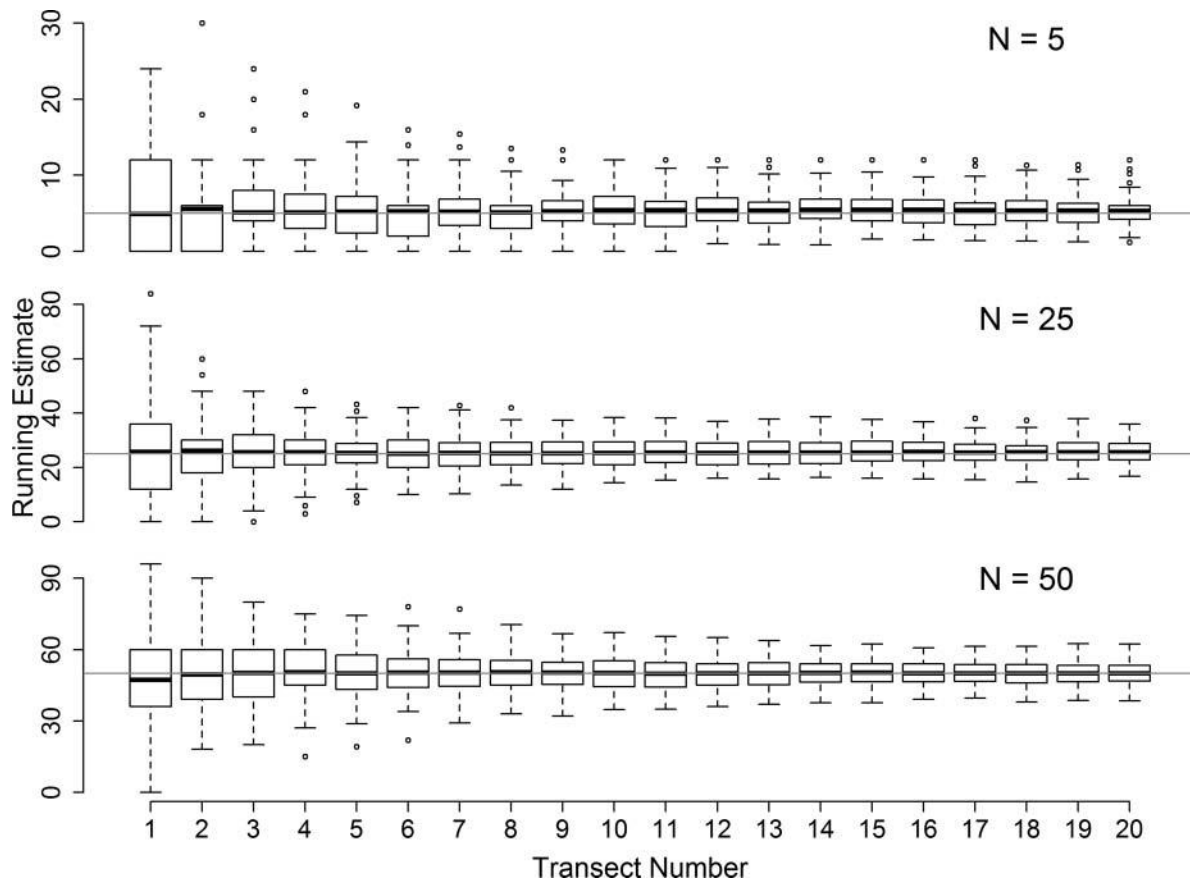
FIGURE 2.   Precision and bias of the DIDSON-based abundance estimate are a function of the number of transects and sturgeon density. One hundred site visits were simulated, each consisting of 20 transects within a 100-m × 300-m stream unit with 5, 25, and 50 uniformly distributed sturgeon. The mean (box center line) and first and third quartiles (box dimensions) of the 100 estimates/transect are plotted, with whiskers extending 1.5 times the interquartile range. Outliers are plotted as separate dots outside the whiskers. The true number of fish in each simulation is shown as the gray line, indicating that the mean number estimated equals the true number of fish.

all within the 95% confidence intervals of the habitat-unit-specific mark–recapture estimates. The three exceptions are the upper limit of the DIDSON estimate at unit 15, the lower limit of the DIDSON estimate at unit 39, and the lower limit of the DIDSON estimate at unit 60. The DIDSON-based estimate of the total and the 95% confidence interval of this estimate is also within the 95% confidence interval of the mark–recapture estimate. Additionally, the confidence interval of the mark–recapture estimate was generally wider than that of the DIDSON method for the habitat-unit-specific and total estimates.

DISCUSSION

The DIDSON-based method of abundance estimation improves, in at least two ways, upon traditional methods that require the capture and handling of individuals. First, this method avoids the negative side effects associated with handling individuals by remotely sensing their presence. This has greatly influenced the ability of researchers to monitor SDPS

spawner abundance without the hazard of disturbing spawning aggregations or inducing unnecessary stress related to capture and handling. In mixed species cases for which the capture and handling of individuals should be avoided, such as in the SDPS, species proportions can be estimated using underwater video camera transects (Groves and Garcia 1998). Green and White sturgeons are easily distinguished using an underwater video camera due to their morphological differences, such as the presence and number of scutes, the color, and the patterns of coloration (Moyle 2002). By not handling individuals, researchers may greatly reduce the timeline for field sampling to occur as this "hands off" method is typically exempt from permitting requirements. Second, this method produces an accurate and cost-effective way to evaluate the abundance and distribution of Green Sturgeon. Our comparison showed that, with greatly reduced effort, the DIDSON transect-based estimator produced superior confidence intervals when compared with the mark–recapture framework. While we did not calculate the empirical probability of the ability of the DIDSON to detect sturgeon within a habitat unit, we suspect it is much
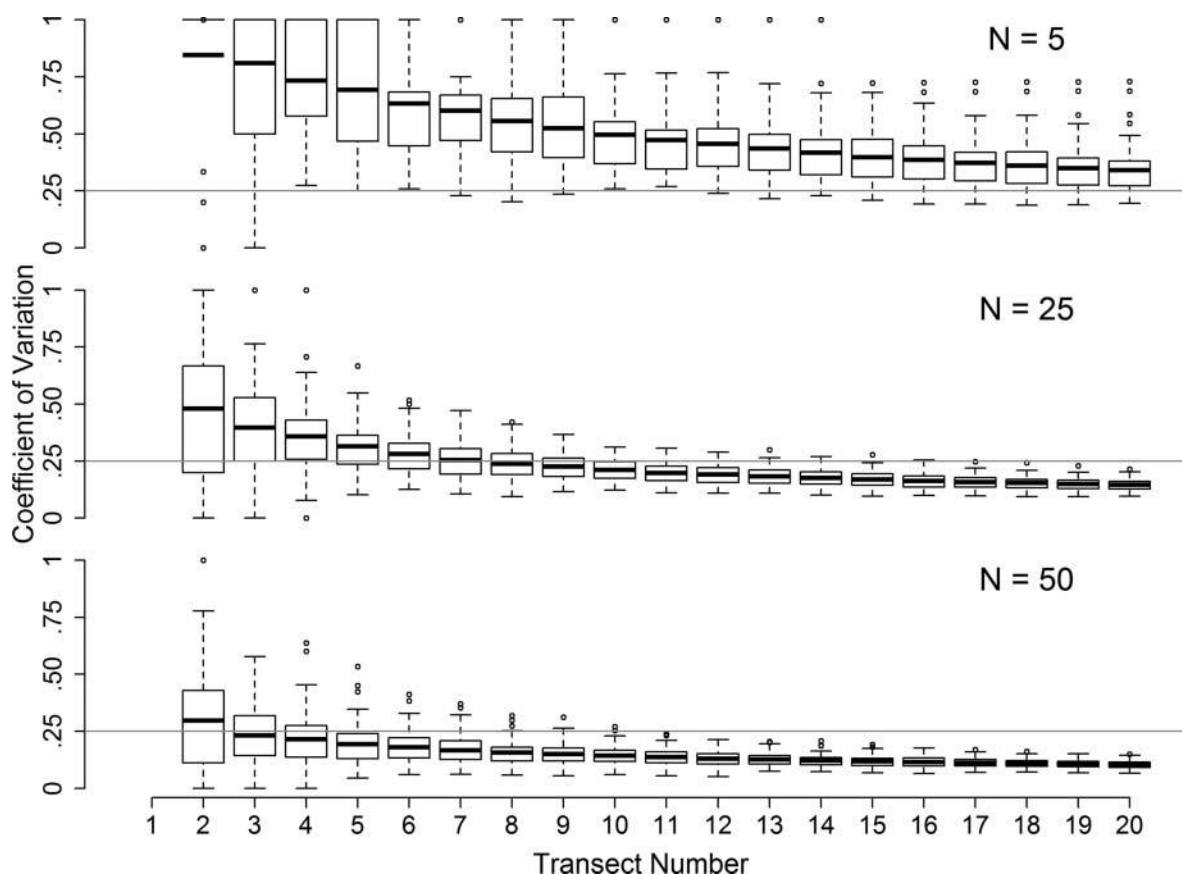
FIGURE 3. Coefficient of variation of the DIDSON-based abundance estimate is a function of the number of transects and sturgeon density. One hundred site visits were simulated, each consisting of 20 transects within a 100-m × 300-m stream unit with 5, 25, and 50 uniformly distributed sturgeon. The mean (box center line) and first and third quartiles (box dimensions) are plotted, with whiskers extending 1.5 times the interquartile range. Outliers are plotted as separate dots outside the whiskers. The gray line shows a coefficient of variation of 0.25. Here the coefficient of variation of the estimated total number of sturgeon drops below 0.25 after a feasible number of transects when $N = 25$ or $N = 50$. The estimation technique is less precise when $N = 5$.

higher than that of the capture techniques used in mark–recapture estimation. This is due to the mobile nature of DIDSON transects and the ability of the field technicians to rotate the DIDSON and "search" for sturgeon during the presence–absence surveys. (It should be noted that during abundance estimation transects, the DIDSON should be pointed along the path of the boat to ensure that the GPS path represents the viewed area of the DIDSON.) Intrinsically, it would be much

TABLE 1. Results of DIDSON abundance estimations at each habitat unit where sturgeon were detected. Total sturgeon is an estimate of the total number of sturgeon detected with DIDSON, regardless of species. Green Sturgeon is an estimate of the total number of Green Sturgeon detected with DIDSON after incorporating an estimate of species proportion; N is an estimate of the total number of sturgeon at each habitat unit.

Unit number and species	Number of transects	N	SD	Lower 95% CI	Upper 95% CI
1	3	20	6	−7	47
15	5	70	11	39	100
24	6	7	1	5	9
35	4	7	3	−1	15
39	7	21	4	11	30
44	5	6	2	1	11
53	3	34	8	−1	69
57	4	24	2	17	29
60	3	38	15	−27	104
Total sturgeon		226	22	181	270
Green Sturgeon		223	22	180	266

TABLE 2. Delta AIC_c and AIC_c weights of the four models implemented to estimate Green Sturgeon abundance using the program MARK.

Model	Delta AIC_c	AIC_c weight
Time-varying p = time-varying c	0.00	0.979
Constant p = constant c	8.51	0.014
Constant p, constant c	10.44	0.005
Constant p, time-varying c	12.09	0.002

more time intensive, costly, and physically invasive for the mark–recapture method to achieve abundance estimates of a similar precision as DIDSON-based transects.

A CV of 0.25 was an arbitrarily chosen level of precision to relay confidence in our final abundance estimate. In comparison, a DIDSON-based estimate of the density of common jellyfish *Aurelia aurita* using an unspecified transect estimator resulted in CVs of 1.25 and 1.70 (Han and Uye 2009). In contrast, a stationary deployment of a DIDSON camera at a salmonid counting station resulted in a CV of 0.14 (Cronkite et al. 2006). Thus, we feel that our reference CV near or below 0.25 to be a balance of what is achievable in the field and what is a useful result for the management of this species.

At most locations we performed fewer transects than what our simulations would suggest was optimal. This practice was sufficient to get an accurate estimate of sturgeon abundance in the habitat units. The few habitat-unit-specific estimates for which the lower bounds of the confidence intervals resulted in negative values were the result of a low number of transects. This could have been remedied by performing a greater number of transects during these surveys (Figure 2). Initially we intended to use a bounded-counts estimator in combination with DIDSON transects to estimate the number of sturgeon present at each location. Upon further investigation we determined that this approach would violate key assumptions of the bounded-counts method (Routledge 1982; Seber 1982).

Specifically, it was not theoretically possible to count all animals on a single occasion (transect) as the habitat unit was much wider than the field of view of the DIDSON camera. Thus, our simulations show the reductions in transect numbers to be a sacrifice in precision.

The DIDSON-based estimates may not be without bias, however. Any violation of the listed assumptions would result in a bias of the final abundance estimate. Two of the assumptions for the DIDSON-based method, assumptions 1 and 5 from above, relate to the movement of individuals. Thus, it is important to supplement DIDSON-based studies with individual-based movement rates from tagged fish (acoustic tags, radio tags, etc.) to estimate sturgeon movement patterns during the sample periods. We suspect that our results were not biased by the movement of individuals as Green Sturgeon typically exhibited small home ranges during our study period (Erickson et al. 2002).

In the future it will be possible to correct the DIDSON-based estimate for the bias induced by moving individuals by using information from tagged fish. Within the spawning grounds of the SDPS, researchers currently operate an array of over 300 acoustic-tag-detecting monitors (Heublein et al. 2008; Sandstrom et al. 2012) and have surgically implanted acoustic tags into many (>300) Green Sturgeon in either the Central Valley or the mixed-stock Columbia River estuary. Currently the Yurok Tribe fishery group operates an array of acoustic-tag-detecting monitors in the Klamath River (B. McCovey, Yurok Tribal Fisheries Department, personal communication), yet no tag detecting monitors have operated in the Rogue River since the studies of Erickson et al. (2002) and Erickson and Webb (2007).

The assumption that all the locations where sturgeon are present are surveyed, assumption number 4, is best fulfilled by establishing defensible criteria to identify and define the sample units (i.e., all locations greater than 5 m deep [Erickson et al. 2002; Thomas et al. 2014]). Then, perform a survey of the study area to identify the locations that satisfy the criteria

TABLE 3. Mark–recapture estimates of Green Sturgeon abundance at each habitat unit where Green Sturgeon were detected; N is an estimate of the total number of Green Sturgeon. We were unable to sample at units 35 and 53 (not sampled [NS]). No recaptures occurred at units 1 or 44, and we were unable to capture any sturgeon at unit 24.

Unit number and total	Marked	Recaptured	N	SD	Lower 95% CI	Upper 95% CI
1	3	0				
15	22	4	42	15	27	94
24	0	0				
35	NS	NS				
39	16	2	35	18	20	108
44	2	0				
53	NS	NS				
57	11	1	34	29	15	165
60	22	2	73	44	34	241
Total	76	9	236	66	150	424

before sampling with DIDSON. Our study area does omit the region of the Rogue River between Blossom Bar and Rainie Falls, a region accessible to Green Sturgeon yet inaccessible to powered jet boats due to Wild and Scenic Rivers protection. Thus, our study is presented as a comparison of two abundance estimation techniques within an accessible study area and not a run-size estimate for Rogue River Green Sturgeon during 2007.

Assumption number 2 (100% detection in the DIDSON camera field of view) is best fulfilled by the use of multiple trained viewers and estimating the precision and bias of their counts, similar to the methods implemented in the age and growth literature (Evans and Hoenig 1998). The problem posed by estimating the true, yet unknown, number of growth rings in fish tissues is similar to our challenge of estimating the true, yet unknown, number of sturgeon that passed within the field of view of the DIDSON camera. We did not explore the impacts of viewer bias on our results; however, the opportunity exists to measure how susceptible DIDSON-based estimation is to viewer count variation (Evans and Hoenig 1998; Holmes et al. 2006).

Assumption 3 (the calculated densities are unbiased) is best managed through the use of accurate measurements of the area (s) sampled. That process should involve accurate measurements of the transect paths using an appropriate GPS, the careful estimation of the sampled area per transect as shown above, and the use of GIS to calculate the total sampled area. Our study implemented these guidelines. Thus, we suspect our final estimate of the total number of Green Sturgeon in our study area to be the least biased by violations of this assumption.

Our mark–recapture abundance estimates appear to be defensibly implemented in light of the assumptions of this method. It is unlikely that assumption number 1 (a closed population) was violated as Erickson et al. (2002) displayed that Green Sturgeon are not immigrating into or emigrating from the study area during this time. It is possible, but unlikely, that assumption number 2 was violated for our habitat-unit-specific estimate. If sturgeon moved between units between our sampling occasions, this would also bias our estimate of the total number of sturgeon at each location. However, all recaptures occurred in the same location as the initial marking. Our strategy of aggregating the detection histories into three sampling occasions served to address this assumption for the estimate of the total number of Green Sturgeon in the study area. Our estimates appear to be robust against assumption 3 (marking individuals does not affect their probability of recapture) because the two models with the greatest weights both contain the probability of capture (p) being equal to the probability of recapture (c). Finally, it is the least likely that assumptions 4 and 5 were violated as we double-tagged the captured sturgeon.

We will avoid speculating as to why the selected model, time-varying p = time-varying c, contained a time-varying component. This fact suggests that an unknown factor was either increasing or decreasing the probability of capture and recapture during each of the sampling occasions, potentially increasing or decreasing the final estimates of abundance. The presence of this unknown factor supports our results that the DIDSON-based estimation technique may be better suited to estimate Green Sturgeon abundance.

The estimates of the number of annually spawning adults, population size, and demographic structure of each population of Green Sturgeon will be useful for the management of the species. Previously, no efforts were being implemented to gather this information. Our results establish the ability to estimate the number of annually spawning adults. To expand the utility of this method, it would be feasible to combine this method with estimates of spawning periodicity (Erickson and Webb 2007) or estimates of demographic structure of the populations (Beamesderfer and Simpson 2007) to produce estimates of population size. Further, it would be best to empirically measure the demographic structure of spawning adults via length measurements using DIDSON, a method shown to be feasible by Hightower et al. (2013).

The fine-scale locations of detected sturgeon resulting from this method can be used in future habitat assessments of Green Sturgeon. Following multiple presence–absence surveys, unit-level occupancy rates will emerge (Mackenzie et al. 2006) because it is likely that the same spawning and holding sites will be occupied year after year, allowing for interannual variation (Bemis and Kynard 1997). Additionally, presence–absence surveys can be expanded to estimate how frequently habitats shallower than 5 m are occupied. Once patterns of habitat use are identified, that information will be useful to evaluate the degree to which each population is susceptible to spatially correlated catastrophic risk (toxic spills, landslides, poaching, etc.), improving the development of potential management scenarios.

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