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Small roped closures held more harvestable Cyclina sinensis biomass in a Taiwanese wetland open to public clam digging

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Abstract

Public clam digging continues in many coastal wetlands also managed for conservation, so managers need evidence on whether small, low-cost access restrictions retain harvestable stock. At 16 paired stations in Siangshan Wetland, Taiwan, we compared the clam *Cyclina sinensis* (Gmelin, 1791) between plots open to public digging (U) and 6-m^2 closures marked with rope and signs. One closure per station (C1) was excavated after four to five months and again six months later; a second (C2) was excavated once, after 11 months. The primary endpoint was shell-on wet biomass of clams > 30 mm, the approximate size kept by collectors. Closures held about three times this biomass on both dates (model-estimated ratios 3.25–3.55; all lower 95% limits ≥ 1.60). Total density was 2.0–2.9 times higher, whereas the proportion > 30 mm did not differ detectably, indicating that closures held more clams rather than a larger share of large clams. Six months after its first excavation, C1 again held 69 clams > 30 mm, more than the 60 removed, an observation relevant to rotational closures. Power simulations showed that 16 paired stations give 86% power to detect a threefold difference but 42% for a twofold difference; 80% power for the latter would need more than 32 stations. Because closure positions were fixed, nonrandom and not sampled before installation, initial stock differences remain an alternative explanation, and the data do not separate retention, growth and immigration. Baseline-matched, randomized trials with marked clams are the next step in evaluating small closures as a wetland management tool.

Introduction

Public clam digging takes the catch from the same sediment in which the remaining clams live. Digging removes the intended catch and can disturb animals left in the sediment, making spatial restrictions relevant to both stock conservation and habitat management (Brown and Wilson 1997; Spencer et al. 1998). Managers need evidence about the quantity of catch-sized animals present in patches that can be marked and protected while collection continues nearby. Shell-on wet mass above the approximate size kept by collectors provides one practical standing-stock endpoint, hereafter "harvestable biomass". The term describes an operational measure, not a legal minimum size.

Spatial protection is often associated with greater biomass and density within marine reserves, although responses vary among organisms and settings (Lester et al. 2009). A synthesis comparing no-take and partially protected areas found stronger density responses under no-take protection (Lester and Halpern 2008). A closer ecological analogy comes from hand-collected bivalves in Solomon Islands mangrove flats, where permanently and temporarily closed areas held more clams than open areas (Aswani et al. 2015).

Many small-scale fisheries instead open closures periodically, alternating protection with extraction. Studies of community reef fisheries in the Solomon Islands provide a governance precedent for this approach: outcomes depend on opening schedules, extraction pressure and local rules (Cohen and Foale 2013; Cohen et al. 2013). A closure that is excavated and then reclosed represents one extraction-and-protection cycle, although research removal differs from a public fishery opening.

The effects of collection can extend beyond the largest retained animals. Brown and Wilson (1997) experimentally examined disturbance associated with commercial hand digging for clams and baitworms on Maine mudflats and found reductions in several infaunal taxa. Mechanical clam harvesting changed benthic communities, followed by recovery (Spencer et al. 1998). Experiments with cockle hand-raking and recreational clam digging likewise showed disturbance effects on infauna, with responses depending on the taxa and disturbance regime (Kaiser et al. 2001; Griffiths et al. 2006). Effects depend on gear: two hand-operated cockle-harvesting methods had small immediate community effects in Portugal, but the greater efficiency of one gear prompted recommendations for quotas and closed areas (Leitão and Gaspar 2007). These studies suggest two expectations for exposed sediment. If collectors take mainly large clams, the fraction above their targeting size should decline. If digging disturbance or other demographic processes act across sizes, abundance may differ while proportional composition stays similar.

The venerid *Cyclina sinensis* (Gmelin, 1791) is harvested in Taiwan (Hsu et al. 2021). In Gomso Bay, Korea, habitat-suitability modeling identified upper-flat areas as potential habitat using sediment properties and exposure time (Zeon et al. 2022). On Chinese culture flats, groups initially 23–30 mm long reached 36–39 mm over about 13 months (Yu et al. 1996), suggesting that growth is unlikely to replace clams removed above harvest size within a few months. At Siangshan Wetland, we repeatedly saw collectors digging near the stations and keeping clams that we judged to exceed about 30 mm in maximum shell length.

On an intertidal flat, a small roped patch offers a physically permeable boundary within which digging can be restricted. We used paired excavations at 16 stations to address three questions. First, did roped closures hold more harvestable biomass than unfenced plots? Second, did that contrast accompany greater density or a larger proportion of large clams? Third, how much harvestable biomass was present six months after a closure had been excavated? We used the monthly unfenced series to place the comparisons in temporal context and estimate monitoring replication. The design compares standing stock at two dates and does not estimate fishery yield, spillover or the demographic contribution of each process.

Materials and methods

Study area and management setting

Siangshan Wetland is a nationally important wetland of about 1,768 ha on the Taiwan Strait coast of Hsinchu City, northwestern Taiwan, comprising estuarine, saltmarsh, mangrove and intertidal mudflat habitats with diverse benthic assemblages (Chen et al. 2018). Tides are semidiurnal, with a mean range of about 3.3–3.5 m and maxima near 5 m (Ministry of the Interior 2018, p 12). The wetland largely overlaps the Hsinchu City Coastal Wildlife Refuge. Under the zoning in force during sampling, entry and collection in the core zone required a permit, whereas regulated existing fishing continued in the buffer and sustainable-use zones (Hsinchu City Government 2024, Appendix 1, pp 65–66). All stations lay

outside the core zone, and we repeatedly observed public digging around them. The current refuge plan lists *C. sinensis* among taxa that registered fishers may collect (Hsinchu City Government 2024, pp 53–55). We did not record individual collectors' permit status.

Closures were installed in late July 2020, and plots were sampled from August 2020 to September 2021. We selected 16 stations (CW1–2, FC1–4, HM1–4 and HS1–6; Fig. 1) from field observations and interviews with collectors about places they recommended or regularly dug, so that stations covered areas in active use. Placement was therefore not random. Each station had two fixed closures (C1 and C2), each 2 × 3 m (6 m^2) and about 5 m apart. Each month, we excavated one unfenced plot of the same size (U), exposed to public digging, at a previously unexcavated position about 5 m outside the closures. Neighboring plots appeared to occupy similar sediment, but we did not measure grain size or elevation relative to a tidal datum. Station was the paired unit of replication. Closure positions are mapped in Fig. 1; U coordinates were not recorded.

Each closure had four corner bamboo posts, each bearing a sign, and two five-strand scout ropes encircling the perimeter at 60 and 120 cm above the sediment surface. There were no mesh walls or roof. The ropes and signs marked the prohibition on public digging. We observed no digging incursions inside the marked boundaries, and the installations required no maintenance. Closure locations remained fixed.

C1 was first excavated in December 2020, approximately four to five months after installation, then reclosed until June 2021. C2 was first excavated in June 2021, after approximately 11 months of closure. December therefore provided paired C1/U observations and June paired C1/C2/U observations. C1 and C2 were analyzed as separate treatments. August U observations were an early postinstallation reference. The closure plots had no matched preinstallation census.

Collection and measurements

Field access and collection were authorized under a research project approved by Hsinchu City Government (HCG109-0093). All *C. sinensis* samples, from monthly U plots and closures alike, were collected in the same way: we excavated the full 6-m^2 plot to about 10 cm, searched the sediment visually without a sieve and removed every clam found. For each clam, we recorded maximum shell length and shell-on wet mass. Visual searching can miss small or deeply buried animals, but clams below 10 mm were also collected, so we applied no lower size threshold. A plot was assigned a zero catch only when we had confirmed that it was sampled and that the record was complete. Otherwise, the record was treated as missing. The principal comparisons used the complete December and June datasets. Two data-entry corrections are documented in Online Resource 2.

Collectors at the stations kept clams that we judged to exceed about 30 mm in shell length, and we used this size as the approximate targeting threshold. We repeatedly saw people digging and fresh digging

traces near the stations, but we did not count collectors or record their effort or catch, so harvesting intensity was not quantified.

Data structure and endpoints

The primary endpoint was harvestable biomass, the sum of shell-on wet weights of individuals strictly > 30 mm per plot, reported as g m^{-2}. Shell material is included: this is standing stock rather than soft-tissue mass, annual yield or secondary production. Secondary endpoints were total count, total biomass, count > 30 mm and the grouped counts above and at or below 30 mm. Each sampled station-treatment plot contributed one record, including confirmed zero catches. Proportions were undefined for empty plots. Individual lengths were used for distributions and quantiles.

We calculated densities above 26.28 and 26.85 mm, the male and female 50% maturity-reference lengths reported from western Korea by Chung et al. (2011). These are external size thresholds, not measurements of maturity or reproductive output in Siangshan. Rounding sensitivities and complete threshold results are in Online Resource 1.

Treatment models and uncertainty

We analyzed December and June separately with generalized linear mixed models that included treatment as a fixed effect, a random intercept for station and an offset for log plot area. Harvestable and total biomass were modeled with log-link Tweedie distributions, which accommodate both zeros and positive continuous values, and counts with log-link negative-binomial distributions. The proportion of clams > 30 mm was modeled as grouped large and small counts with a beta-binomial distribution, using nonempty plots only. All models were fitted in glmmTMB (Brooks et al. 2017).

The three primary comparisons were December C1/U, June C1/U and June C2/U. We applied the Holm (1979) adjustment to the three harvestable-biomass tests and, separately, to the three tests of each secondary endpoint. Effects are reported as conditional treatment ratios, or odds ratios for proportions, with unadjusted 95% Wald intervals; only p values were adjusted. We checked optimizer convergence, Hessian status and gradients. Diagnostics, including the Poisson-limit fits used for two secondary December thresholds, are given in Online Resource 1.

To check model fit, we compared observed zero counts, variance and maximum biomass with simulations from the fitted primary models. As sensitivity checks, we used paired Wilcoxon signed-rank tests and a station bootstrap those resampled stations with their treatment pairing intact (2,000 draws; seed 20261103). The bootstrap estimates ratios of total biomass or count, which differ from the conditional model ratios, so these are labeled as bootstrap ratios in the Results. We also compared June density of *Parilaternula certa*, a co-occurring clam not targeted by collectors whose recruitment at Siangshan was described by Chang et al. (2027), among C1, C2 and U with the same count model, as a limited check for location or installation effects on a non-target bivalve (Online Resource 1).

Size structure and closure excavation history

Length distributions were summarized by the 10th, 25th, 50th, 75th and 90th percentiles. Pooled treatment-quantile differences were evaluated with station-resampling intervals. Individual-level Kolmogorov–Smirnov tests were descriptive because their conventional form assumes independent animals. The beta-binomial model tested the large-size fraction at station level. Raw range was omitted as an inferential endpoint because it increases mechanically with sample size.

For C1, we compared June with December stock by station bootstrap and compared C1 with C2 within June. Endpoints included total density, harvestable biomass, large-clam counts and the fraction below 15 mm. A two-date C1/U model included treatment, date and their interaction, station and station-by-date intercepts, and the area offset. The treatment-by-date ratio of ratios assessed compatibility between the two contrasts rather than equivalence.

Unfenced series and prospective monitoring

The monthly series used the same 15 stations throughout, excluding HS2 because its August attribution and complete catch could not be resolved. April 2021 was excluded because most records were lost. Thirteen months from August 2020 to September 2021 had complete counts, totaling 1,075 animals. Nine had complete weights. January–March and May 2021 were omitted from biomass summaries because weights were incomplete. The temporal figure uses these same 15 stations for U and closure points. Principal treatment models use all 16. Means and pointwise 95% station-bootstrap intervals used 2,000 draws (seed 20261106), with missing months shown as gaps.

Exploratory calendar trends used a month index, station and month intercepts, and a log-area offset. Biomass used a Tweedie model and counts a negative-binomial model, with station-only intercept fits as sensitivity checks. Following the general simulation approach to GLMM power analysis (Johnson et al. 2015), monitoring simulations used paired unfenced and closure plots at 8, 16, 24, 32 and 48 stations under the June biomass model's fitted variation. True ratios were 1, 1.5, 2 and 3, with 500 refitted simulations per scenario. Failed fits counted as non-rejections. Power was empirically calibrated against the simulated null distribution at each station number. Dispersion sensitivities and simulation uncertainty are in Online Resource 1.

Exploratory growth scenarios

To assess the length increments compatible with the four-to-eleven-month closure periods, we fitted exploratory nonseasonal length-frequency growth curves to the monthly U series using bootstrapped ELEFAN (Mildenberger et al. 2017; Taylor and Mildenberger 2017; Schwamborn et al. 2019). Bin widths, moving averages, search bounds, optimizer budgets and sensitivity results are in Online Resource 1. We calculated elapsed calendar time between specified lengths. The curve anchor was not interpreted as a birth or settlement date.

Software and artificial intelligence assistance

Analyses were run in R 4.6.1 (R Core Team 2026) with glmmTMB 1.1.14, and in Python 3.10. Growth reconstruction used TropFishR and fishboot (Online Resource 1). Correction records, output summaries, random seeds and code reproducing the primary models are provided in Online Resource 2. OpenAI Codex assisted with data auditing and code polishing. Claude assisted with language editing and manuscript review. The authors designed the study, verified all corrections, analyses and text, and take full responsibility for the content.

Results

Greater harvestable biomass in the closures

Closures contained more harvestable biomass at both comparison dates (Table 1; Figs. 2–3). The fitted December C1/U ratio was 3.25, with a 95% interval of 1.60–6.60. June ratios were 3.55 for C1/U, with an interval of 1.76–7.16, and 3.30 for C2/U, with an interval of 1.63–6.70. Holm-adjusted *p* values were 0.0019, 0.0012 and 0.0019, respectively.

Station-based sensitivity checks supported the same direction. Bootstrap ratios of total harvestable mass were 3.38 in December and 3.39 and 3.08 in June, with respective 95% intervals of 1.62–6.60, 1.68–10.13 and 1.53–11.78. All three paired Wilcoxon comparisons were significant after Holm adjustment, with adjusted $p = 0.0176$. The December and June C1/U contrasts were compatible: the estimated June/December ratio of treatment ratios was 1.07, with a 95% interval of 0.43–2.68 and interaction $p = 0.89$. The repeated dates thus showed a common direction and similar central estimates for the primary endpoint.

Table 1

Observed stock at the 16 stations. Each station-treatment plot covered 6 m^2, giving 96 m^2 per treatment-date. Biomass includes shells. Harvestable mass includes individuals strictly > 30 mm; percentages are pooled individual proportions. Conditional model effects are shown in Fig. 3

Month	Plot	Total n	Density (m^{-2})	Total mass ($g\ m^{-2}$)	Harvestable mass ($g\ m^{-2}$)	> 30 mm (%)
2020-12	C1	255	2.66	16.71	9.19	23.5
2020-12	U	87	0.91	5.69	2.72	24.1
2021-06	C1	387	4.03	18.66	8.56	17.8
2021-06	C2	284	2.96	15.52	7.78	21.5
2021-06	U	134	1.40	6.76	2.53	14.2

Density and size composition

Total density was 2.0–2.9 times higher in closures (Table 1). Negative-binomial ratios were 2.86, 2.88 and 2.02 for December C1/U, June C1/U and June C2/U, and all three were significant after Holm adjustment. Total biomass followed the same pattern.

The station-level model detected no difference in the proportion > 30 mm (Table 1): closure/unfenced odds ratios were 1.26, 1.49 and 1.98, with respective 95% intervals of 0.57–2.78, 0.60–3.68 and 0.79–4.95. Median shell lengths were 24.83 and 25.81 mm in December C1 and U, and 22.20, 24.45 and 23.70 mm in June C1, C2 and U. Densities above both external maturity-reference lengths were higher in the closures. Full threshold and quantile results are reported in Online Resource 1.

June density of the non-target clam *Parilaternula certa* did not differ detectably among C1, C2 and U (density ratios 0.96–1.03, with 95% intervals spanning approximately 0.23–4.33; Online Resource 1).

Stock present after the first C1 excavation

June C1 contained 69 clams > 30 mm, compared with 60 in December C1 and 61 in June C2. Its total density was 1.52 times the December C1 density, with a station-bootstrap interval of 0.95–2.49. The corresponding harvestable-biomass ratio was 0.93, with an interval of 0.46–2.24. Relative to June C2, C1 density and harvestable biomass had ratios of 1.36 and 1.10, with intervals of 0.81–2.26 and 0.56–2.19. The proportion below 15 mm rose from December to June in U (3.4% to 14.2%) and C1 (3.5% to 17.3%). June C2 had a similar proportion (15.1%).

Temporal context and monitoring sensitivity

Monthly unfenced-plot harvestable biomass varied from 0.56 to 5.03 g m^{-2} across the nine months with complete weights (Fig. 4). The means for the 15 stations in August 2020, December 2020 and June 2021 were 2.40, 2.90 and 2.61 g m^{-2}. The exploratory linear biomass trend had a monthly ratio of 1.025, with a 95% interval of 0.964–1.089 and $p = 0.43$. The count series yielded a monthly ratio of 0.994, with an interval of 0.928–1.065 and $p = 0.86$. The closure means plotted at these same stations were higher at the comparison dates.

Under the simulated future design, 16 stations had a calibrated power of 0.858 for a true threefold harvestable-biomass ratio and 0.418 for a twofold ratio. Power for a twofold ratio increased to 0.616 at 24 stations, 0.772 at 32 and 0.852 at 48.

Exploratory growth scenarios

Exploratory nonseasonal length-frequency fits produced setting-specific median K values of 0.123–0.308 yr^{-1} and median L_∞ values of 54.52–60.00 mm, with substantial sensitivity to analytical and optimizer settings (Online Resource 1). Median average-calendar times from 10 to 30 mm ranged from 17.89 to 54.44 months across settings. The corresponding 2.5th percentiles ranged from 11.3 to 31.3 months (Fig. S1, Online Resource 1).

Discussion

Closures held more harvestable biomass

Small roped closures held about three times the harvestable biomass of neighboring unfenced plots on both sampling dates, and model-based, paired-rank and station-bootstrap analyses agreed on this contrast. The difference recurred across 16 stations and two dates, so it was not confined to one locality or sampling occasion. Because the endpoint is set at the approximate size kept by collectors, the contrast measures the part of the stock that public digging removes. Closures also held more clams above the maturity-reference lengths reported for *C. sinensis* in western Korea, although local maturation schedules may differ (Chung et al. 2011).

The direction agrees with the greater abundance of hand-collected bivalves in permanently and temporarily closed mangrove flats in the Solomon Islands (Aswani et al. 2015), here at a much smaller plot scale. Closure responses of harvested bivalves are not uniform, however. Commercially fished and non-fished Australian *Donax deltoides* beaches showed no detectable density decline or size truncation during the harvest season (Gray 2016), and small closed areas for Maine sea scallops showed some size and recruitment differences but no density difference (Cleaver et al. 2026). These surf-beach and subtidal systems differ from Siangshan, but they show that a closure effect cannot be assumed and needs to be measured in the setting where it will be applied.

Alternative explanations and pathways to higher stock

Initial location differences are the leading alternative to protection. Stations were chosen from field observations and collector recommendations, closure positions were fixed, U plots moved each month, and closure plots were not sampled before installation. A location that initially supported more clams could therefore remain different without any effect of protection. Local pairing and visually similar sediment reduce broad habitat differences but cannot separate initial stock from lower removal. Bivalve abundance and size varied with sediment and estuarine setting among Spanish intertidal grounds (Cerdeira-Arias et al. 2024), harvesting history and clam abundance covaried in a Galician comparison (Román et al. 2023), and a habitat model for *C. sinensis* in Gomso Bay used sediment and exposure time to identify potential sites (Zeon et al. 2022). Measuring these variables, and stock itself, before closures are assigned would address this alternative directly.

The closure structure is a second possible source of difference. Mesh cages can alter flow as well as exclude consumers (Miller and Gaylord 2007), whereas our closures had no mesh walls or roof and left the sediment open to water and animals. The bamboo posts could nevertheless serve as bird perches, and flow and sedimentation were not measured. June density of the non-target *P. certa* did not differ detectably among treatments, but its wide intervals and different ecology, including late-winter to early-spring recruitment at this site (Chang et al. 2027), mean that this check cannot rule out location or installation effects (Online Resource 1).

If protection did contribute, the size data indicate how. Collectors kept large clams, yet the proportion > 30 mm did not differ detectably between closures and unfenced plots: closures held more clams across sizes rather than a larger share of large ones. This pattern would follow if digging disturbs animals across several sizes, as reported in other intertidal collection settings (Brown and Wilson 1997; Spencer et al. 1998), or if recruitment, survival or movement change several size classes together. The odds-ratio intervals still permit appreciable proportional shifts. Growth could add to harvestable biomass within closures because clams that cross 30 mm there are not removed. Median times from 10 to 30 mm of about 18–54 months across the exploratory growth settings exceeded the four-to-eleven-month closure periods, so growth of clams already near 30 mm is a more plausible short-term contribution than growth from small sizes. The sensitive, nonseasonal fits do not quantify this contribution.

Reappearance after excavation and periodic closures

Six months after its first excavation, C1 again held 69 clams > 30 mm, compared with 60 removed in December. Because every clam encountered in December was removed, these animals must have escaped the excavation, lain below its approximately 10-cm depth or arrived afterward, and survivors and arrivals could have grown before June. Ma et al. (2025) set culture substrate depths from observed burial to about three shell lengths, but their experiment began with clams averaging 13.44 mm and did not measure adult burial depths in the field, so the fraction missed below our excavation cannot be estimated. The proportion below 15 mm increased between December and June in U and C1 and was similar in C2. Small clams in C1 therefore do not distinguish refilling of the excavated plot from a seasonal change shared across plots.

Movement was not measured at Siangshan, but it is plausible. Ma et al. (2025) described small displacements and repeated burrowing attempts by *C. sinensis* in shallow laboratory substrate. In other venerids, adult *Mercenaria mercenaria* crawl at the sediment surface (Tettelbach et al. 2017), *Austrovenus stutchburyi* crawls farther under bioturbation or macroalgal cover (Mouritsen 2004), stranded *Meretrix lusoria* of mainly 25–40 mm indicate mucous-cord drifting (Takahi and Henmi 2011; Nakamura 2015), and wave-driven bedload redistributes newly settled *Ruditapes philippinarum* of about 1 mm (Toba et al. 2011). These are process precedents, not movement rates for *C. sinensis* or evidence that it drifts on mucous cords. In a 6-m^2 plot, even a few arrivals or departures could change the observed density.

The C1 history resembles one cycle of a periodically harvested closure: protection, extraction and reclosure. In reef fisheries, openings removed much of the biomass accumulated during closure (Goetze et al. 2018), reclosed areas recovered variably and gave no benefit to highly vulnerable species (Goetze et al. 2016), and openings concentrated fishing effort under rules that varied among communities (Cohen et al. 2013). Madagascan octopus' closures of a few months, by contrast, increased catch and catch per unit effort after reopening (Oliver et al. 2015). Outcomes therefore depend on the species and on how much is removed at opening. Those studies measured catch after opening, whereas we measured standing stock after a research excavation that removed every clam encountered. With June

C1/C2 intervals that allowed a broad range of differences, the six-month interval is a candidate for testing rotation schedules rather than an evaluated one.

Scale, zoning and wetland objectives

A 6-m^2 patch has a large boundary relative to its area, so clams and digging effort can cross it easily. Benefits of fish reserves increase with no-take area and protection age (Claudet et al. 2008), although that relationship cannot be transferred directly to small clam plots, and spatial models predict that fishing effort concentrates along reserve boundaries (Kellner et al. 2007). Greater stock inside small patches therefore shows a local response, and its value for a managed wetland depends on placement, boundary exchange and where displaced digging goes. For zoning, the next question is whether larger contiguous closures, or rotating closed zones such as the north–south zones established in Taijiang National Park, where future rotation is linked to annual quantitative surveys of *C. sinensis* (Taijiang National Park Headquarters 2020), hold proportionally more stock than scattered small patches. A larger closure has less boundary relative to its area and could exchange fewer clams with fished sediment, but it could also push more digging onto neighboring flats.

Clam stock is only one wetland objective. In Strangford Lough, experimental cockle raking reduced eelgrass biomass, leading to a recommendation to keep harvesting outside eelgrass beds (McLaughlin et al. 2007). Changes in fished cockle and mussel stocks were associated with oystercatcher survival in the Wash (Atkinson et al. 2003), and oystercatchers in fished Wadden Sea areas had poorer condition despite nearby protected grounds (Verhulst et al. 2004). Whether closures at Siangshan benefit birds would depend on local prey identity, accessible prey sizes and bird use, none of which we measured.

Design limitations

Several constraints have been discussed above: fixed, nonrandom closures without a preinstallation baseline, one plot per treatment at each station, two principal comparison dates, and no direct measures of habitat, harvesting intensity, compliance, flow or post effects. Visual excavation to about 10 cm left recovery efficiency uncertain, especially for small or deeply buried clams. Missing weights and the incomplete April record limit the temporal series, and moving U positions leave any cumulative effect of research excavation unresolved. The growth fits lack local age validation. Within these limits, the repeated contrast supports a local stock association. Demographic attribution, whole-wetland yield, spillover and sustainable rotation periods require the designs outlined below, and the repeated-date and C1/C2 intervals indicate compatibility, not equivalence.

Monitoring and the next management trial

Hsinchu City Government and registered fishers could test temporary roped closures within authorized existing-use areas under the refuge conservation plan. Candidate plots should be surveyed before installation and closures assigned at random within tidal-elevation and sediment strata, with repeated observations before and after installation in replicated closure and reference plots to separate treatment effects from background change (Underwood 1994). Area, excavation depth and recovery methods

should be kept constant. Marked clams and immediate recovery checks would separate movement, survival, growth and missed animals, and comparisons of closure sizes, with surveys beyond closure boundaries, would measure redistribution of both clams and digging. For periodic openings, opening duration, collection methods and extraction limits should be set in advance, and catch, effort, boundary condition, compliance and stock followed through several cycles.

Replication should match the change managers need to detect. Under the variation observed here, 16 paired stations were adequate for a threefold contrast but not for a twofold one, which would require more than 32 stations (Online Resource 1). Stock, catch and collector effort should be recorded separately. In a New Zealand urban cockle assessment, comparisons of harvest with estimated yield were sensitive to recruitment size, biomass change and harvester selectivity, leading the authors to question yield-based management alone (Hartill et al. 2005). Following these quantities through several seasons and closure cycles would show whether the stock contrast persists and how closures can be adjusted to maintain the resource.

Conclusions

Small roped closures held about three times the harvestable biomass of neighboring unfenced plots at Siangshan, with greater clam density on both comparison dates. Six months after excavation, a reclosed plot again held more clams > 30 mm than had been removed. A simple, low-cost restriction on digging therefore warrants consideration as a patch-scale measure in wetlands where public collection continues. Whether the benefit holds for larger contiguous or rotating closed zones, and where displaced digging moves, are the questions that matter for zoning. Replicated trials with baseline surveys and randomized placement, nested within existing refuge zoning, can answer them and give managers a direct basis for deciding where and for how long to close harvesting areas.

Declarations

Compliance with Ethical Standards

Field access and collection were authorized through the research project approved by Hsinchu City Government (HCG109-0093).

Competing Interests

The authors declare that they have no competing interests.

Electronic supplementary material

Online Resource 1 (ESM_1.pdf) Supplementary methods, complete model summaries, exploratory growth sensitivity, calendar-time interpretation and monitoring simulations

Online Resource 2 (ESM_2.zip) Twenty-six machine-readable CSV tables, a primary-model reproduction script, a file inventory and manuscript metadata

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Author Contribution

C.-S.G.C., J.-S.J., A.S.-T. and S.-S.Y. collected field data. S.-S.Y. conceived the study and acquired funding. C.-S.G.C. and J.-S.J. curated the data and performed the formal analysis. C.-S.G.C. prepared all figures. C.-S.G.C. and S.-S.Y. wrote the manuscript. All authors reviewed and approved the final manuscript.

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Data Availability

The plot-level data, corrected individual measurements and analytical summaries supporting the findings of this study, together with a script reproducing the primary biomass models, are included in the supplementary information files (Online Resource 2). The original field records are available from the corresponding author on reasonable request.

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Figures

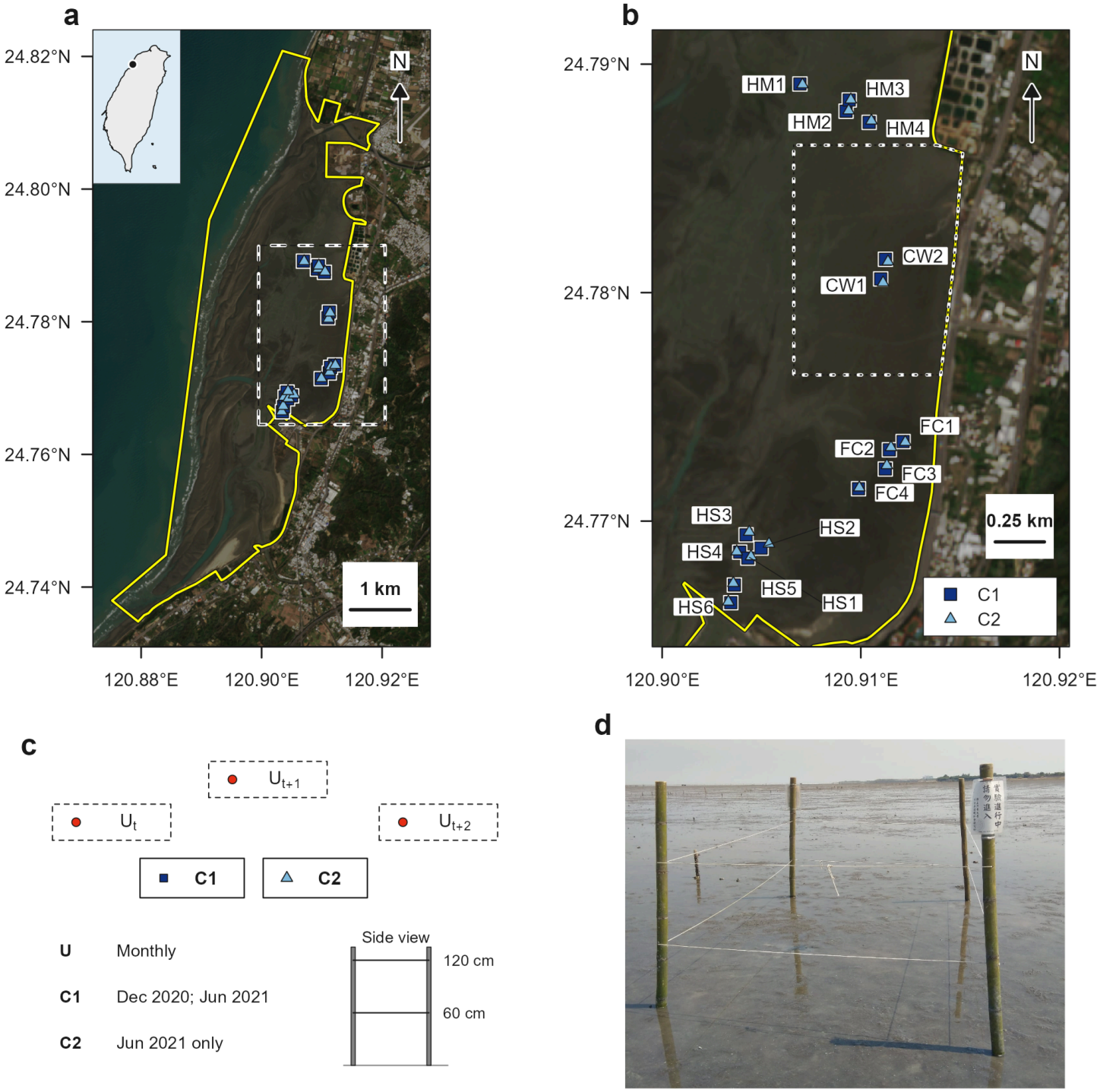


Figure 1

Study location and plot design. (a) Siangshan Important Wetland boundary (yellow), sampling extent (dashed box) and Taiwan locator inset (Natural Earth, 1:10 million). (b) C1 squares and C2 triangles at 16 stations in WGS84; they lie approximately 5 m apart and overlap at this scale. The dotted outline marks the environmental education zone. All study stations lie outside the refuge's core zone. U coordinates and tidal elevations were unavailable. Background: Sentinel-2 L2A, 4 February 2021; contains modified Copernicus Sentinel data 2021 and provides geographical context. (c) Illustrative sampling layout: C1

and C2 remained fixed, while successive monthly U plots occupied different positions around their periphery. U_t, U_{t+1} and U_{t+2} represent example months, not mapped positions or a prescribed sampling direction. Circles denote U, squares C1 and triangles C2. Each plot covered 6 m^2, with approximately 5 m between boundaries; the side view shows the two ropes at 60 and 120 cm above the sediment surface, supported by four signed bamboo posts, without mesh. (d) Field photograph of a roped plot with bamboo posts and a notice (sign reads "Experiment in progress, keep out")

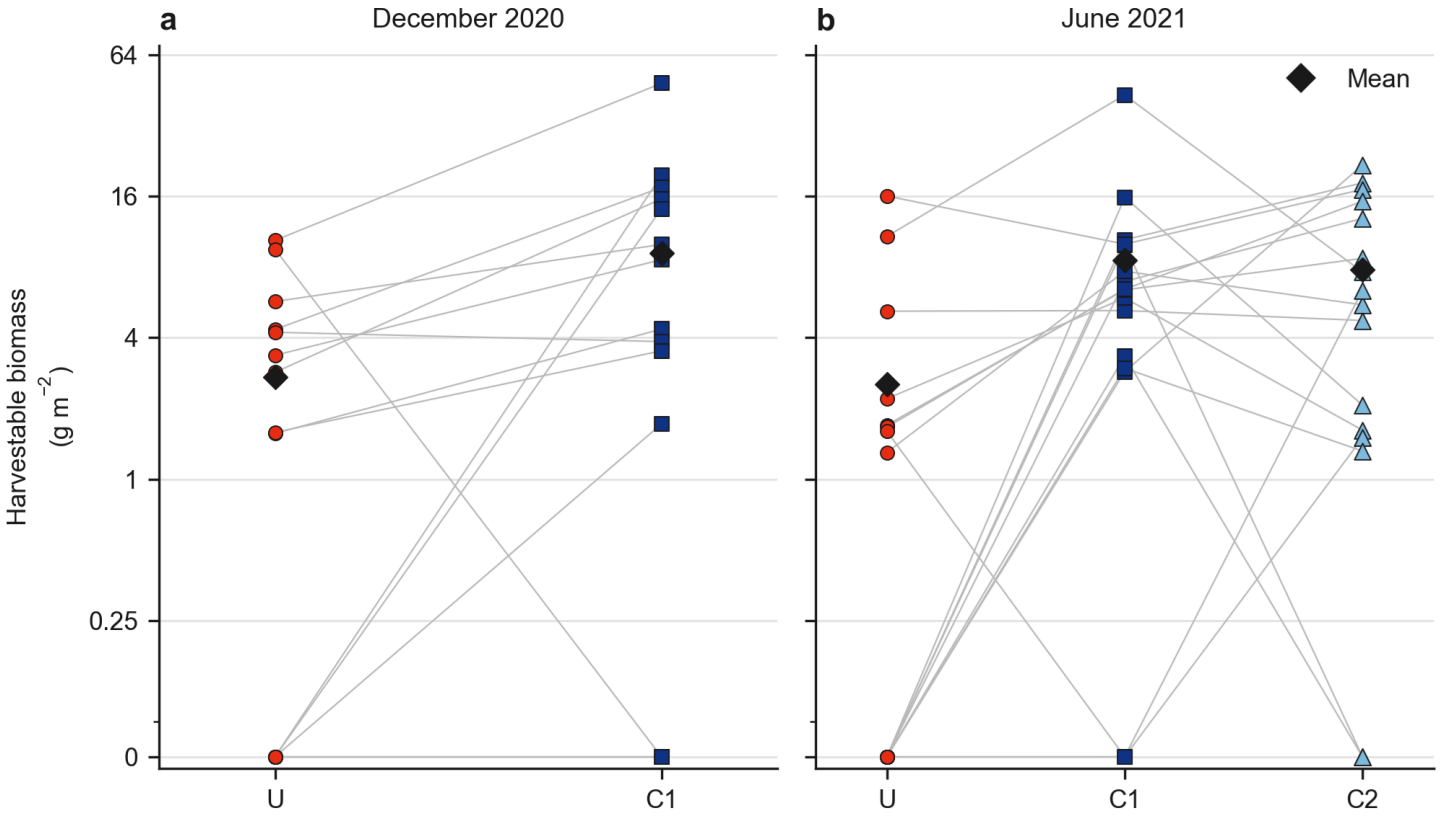


Figure 2

Paired harvestable biomass (>30 mm) at 16 stations in (a) December 2020 and (b) June 2021. Circles, squares and triangles denote U, C1 and C2. Lines connect treatments at each station. The symmetric logarithmic scale has a linear region up to 0.2 g m^{-2}; diamonds are arithmetic means. Biomass is shell-on wet mass per square meter

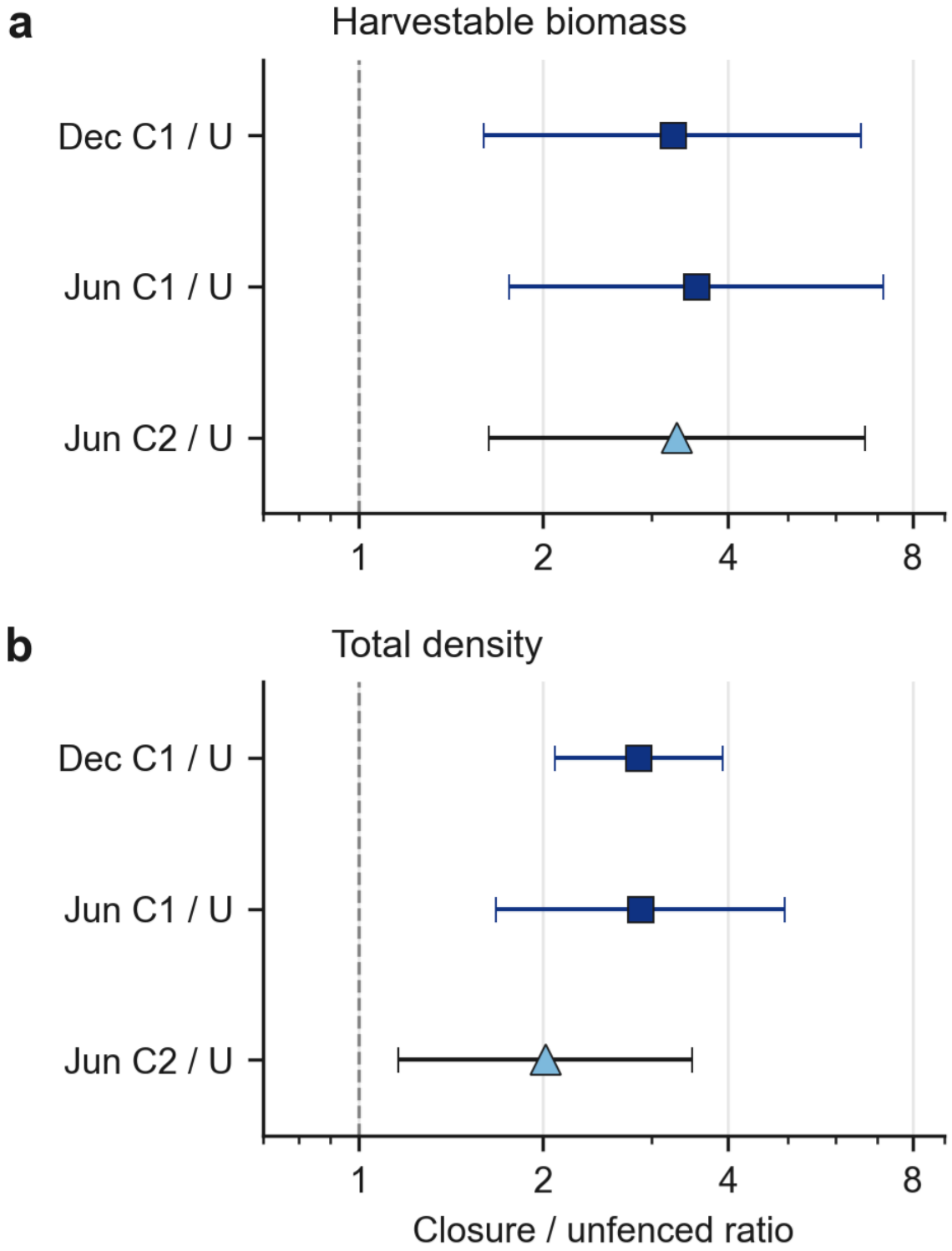


Figure 3

Estimated closure/unfenced ratios for (a) harvestable biomass and (b) total density. Squares (C1/U), triangles (C2/U) and bars show model ratios and unadjusted 95% Wald intervals. The reference line indicates one. All three primary biomass comparisons were significant after Holm adjustment. Counts use negative-binomial models and biomass Tweedie models, with station intercepts and area offsets

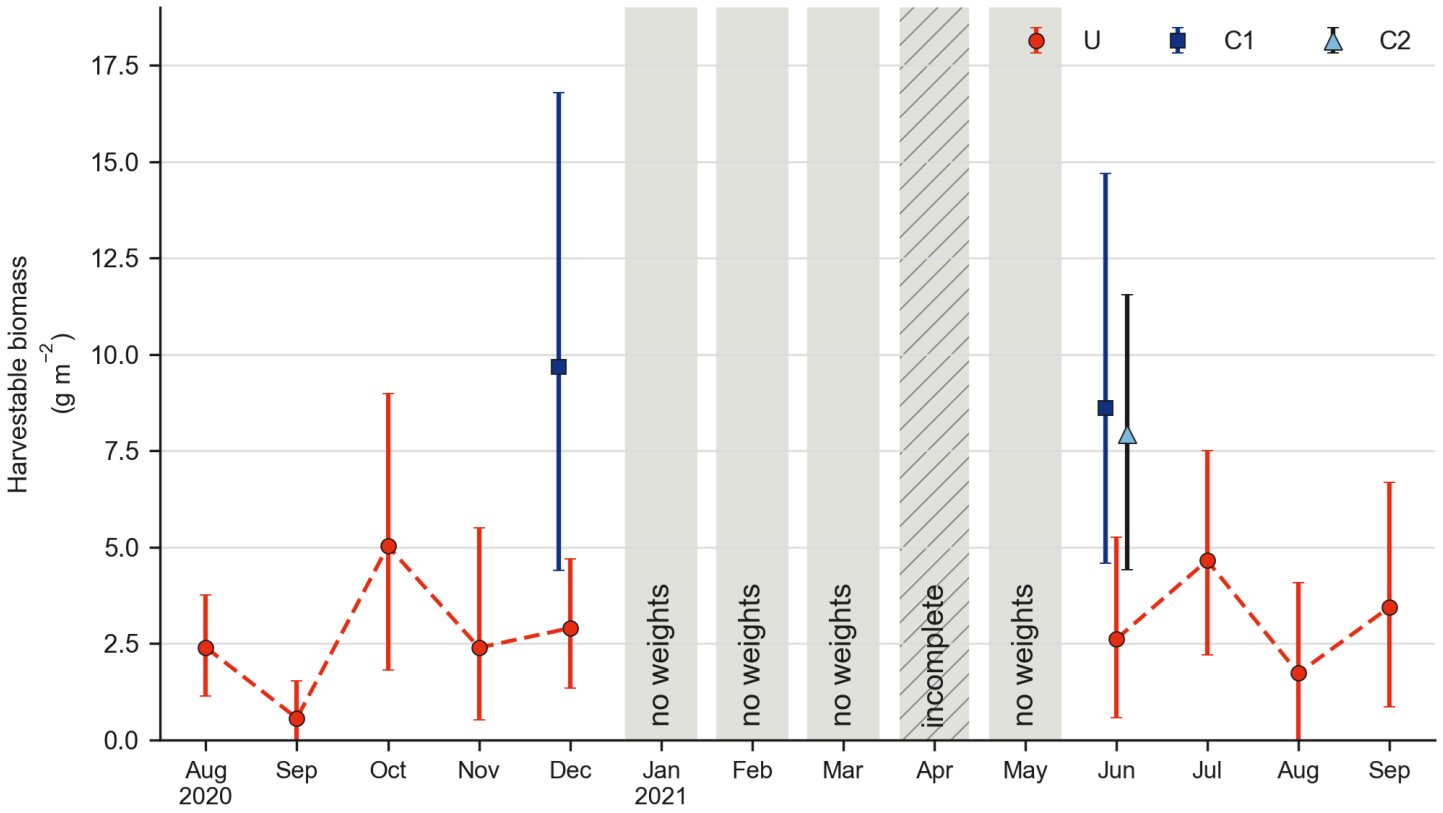


Figure 4

Monthly harvestable biomass in U with same-month closure observations at the same 15 stations. Circles (U), squares (C1) and triangles (C2) are arithmetic means; bars are pointwise 95% percentile intervals from 2,000 station resamples. "No weights" marks January–March and May 2021; "incomplete" marks April. All plotted treatments exclude HS2, whereas the main treatment models use 16 stations; closure means therefore differ slightly from Table 1. August is an early postinstallation U reference. Lines connect adjacent observed U means only

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- ESM1.pdf